Quantifying the effects of Entry Level Stewardship (ELS) on biodiversity at the farm scale: the Hillesden Experiment

Final report

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1 Executive Summary

- The Hillesden experiment has proved to be a valuable, practical test of the effectiveness of the Entry Level Stewardship scheme for a range of taxa at the farm-scale. The experiment was a large scale, randomised block experiment with three treatments applied to replicated land areas of *c.* 50-60 ha: i) Cross compliance (CC, conventional intensive arable management subject to the EU rules of cross compliance), ii) Entry Level Stewardship (ELS, a small suite of options selected to replicate a 'standard' agreement comprising 1% area out of production), iii) Entry Level Stewardship Extra (ELS X, a more extensive selection of options targeted at farmland wildlife of conservation interest comprising 5% area out of production).
- The results show that over a five year period (2006-2011) the effects on (i) habitat quality and food resources, and (ii) the abundance, diversity and population dynamics of some key farmland taxa were significantly enhanced by both targeted local management and enhancements at the landscape scale. However there were clear effects of scale on the responses of different taxa (see Table 1.1). Mobile species like bumblebees, moths, birds and small mammals were more subject to the effects of wider landscape context and showed a net response to the whole farm manipulation. The response of other taxa e.g. plants and some invertebrate groups, were more influenced by local environmental effects and plot level manipulations.
- In general resource quality was higher in the more 'targeted' ELSX options. However there was high variability in results reflecting species specific responses to habitats at both local and landscape scales. Critical to the success of any ELS agreement is the quality of plot establishment and maintenance. Often if a sown option is not doing well after the first year, it will be unlikely to improve and should be abandoned. Both ground conditions at establishment and subsequent weather conditions contributed to the high plot variability observed.
- We suggest that greater emphasis should be placed on creating a much higher proportion of field margin habitat that gives a higher probability of providing a range of critical both winter and summer resources for farmland wildlife. It is not possible to provide these resources within a single margin type, so a diversity of margins is required at the farm scale.

	Indicator Taxa										
	Hedge berry production	Seed production in bird plots	Dicotyledon flower abundance	General invertebrate abundance	Pollinator abundance	Moth abundance	Small mammal abundance	Breeding bird territories	Breeding bird productivity	Winter bird abundance	Tree hole bird productivity (Nest boxes)
Cross Compliance (Control)	↔	↔	↔	↔	↔	↔	↔	↔	↔	↔	↔
Entry Level Stewardship (ELS)	↔	↑	↑	↑	↑	↔	↔	↔	↔	↔	↔
Entry Level Stewardship Extra (ELS X)	↔	↑ ↑	↑ ↑	↑ ↑	↑ ↑	↔	↔	↑	↑	↑	↑
Whole farm	↔	↑	↑	↑	↑	↑	↑	↑	↑	↑	↑

Table 1.1. Summary table of main results for indicator taxa. (↔=little or no change, ↑ =significantly higher than CC, ↑↑=significantly higher than ELS)

- Farm-scale bird abundance (for all species grouped, all granivorous species and seven individual species, including Yellowhammer) increased significantly after the baseline year mainly as a result of increases in food resources and habitat provided by sown seed patches. However these effects were not sustained over the years because of variable environmental conditions i.e. extreme weather; including the two coldest winters since the mid 1990s.
- There was evidence of positive spillover effects of increased food resources on populations of granivorous birds. Birds do not simply move from food-rich patch to food-rich patch across the landscape, but also sample and benefit from, intervening habitat of different types. Thus across a farm/landscape the optimum management would provide a range of seed resources through winter combined with habitats providing high invertebrate abundance in summer. This may best be achieved through the provision of structurally and floristically diverse swards.

- Relaxed (biennial) hedge cutting does not appear to increase berry yield of common hedgerow species (hard and soft fruited berry counts and hard fruit biomass). This option is unlikely to achieve the aim of substantially increasing biomass of berries and provides relatively little benefit to wildlife as food resources for over winter compared to typical (annual cutting).
- Provision of increased floral resources (as in ELS X) enhanced both the general abundance of invertebrates and pollinators like bumblebees, hoverflies, solitary bees and butterflies along with the number of visits they made to flowering plants. Differences in habitat type are important at a local scale in promoting the abundance and diversity of some pollinators, but at the landscape scale, the benefits of implementing ELS are likely to be in enhancing landscape heterogeneity which is important for overall invertebrate diversity.
- Small mammal abundance initially increased with a greater number of species recorded in the ELS X that was linked to increased seed resources and habitat availability. The experiment indicated that ELSX did not perform significantly better than the ELS treatment, although there was strong evidence that these treatments performed better than the CC control in increasing mammal abundance, species richness and biomass in Spring and Autumn.
- Significant increases in mammal abundance, species richness and biomass on the CC control also indicated that a whole farm or landscape-scale effect was achieved without having to place ELS or ELSX treatment margins on all fields. The density of ELS and ELSX margins adopted in the experiment appeared capable of enhancing the mammal communities of neighbouring CC margins.
- There is evidence of additionality for some options e.g. the Bumblebird option (EF2C) includes a range of flowers which provide nectar and pollen resources for invertebrates and seed resources for birds. In many cases the enhancement of habitat/resource quantity and diversity had disproportionate positive effects on key taxa relative to their total areas.
- Despite some generally poor weather conditions during the experiment there is evidence that ELS options can increase the abundance and diversity of key declining

invertebrate groups e.g. Section 41 moths, solitary bee and wasps. This suggests that ELS may have a role in buffering populations from more extreme environmental conditions.

- It is clear that for many taxa there are lag times in their responses to treatments. This study highlights that the small, five year time-scale over which ELS currently operates may be insufficient to start observing significant benefits in population abundances e.g. insects that are often have cyclic dynamics and greatly affected by climate, birds that utilise wide spatial scales. In addition, since colonisation time varies across species (due to variation in dispersal ability) and also depends on the surrounding habitat context, it may take many species a long time to colonise the newly available suitable habitats created by ELS unless landscape connectivity is also improved.

2 Summary Guidance for Advisors

- Our results suggest that ELS, as designed and currently implemented, may have limited success at increasing the abundance of farmland wildlife of conservation concern. To improve the scheme's general impact and effectiveness attention must be paid to: i) increasing the areas taken out of production, ii) improving the quality of delivery of options (through improved establishment and maintenance regimes) and adherence to existing criteria, and iii) promoting greater levels of habitat heterogeneity at both the farm and landscape levels through selection of a wider range of options.
- The scheme (and option selection on a particular farm) should consider the landscape at greater than the farm scale. Many of the key farmland taxa utilise habitats across the wider landscape thus maintenance (or growth) of their populations requires resources to be available at this scale. If key resources (e.g. pollen and nectar for bees) appear to be limiting at these wider scales, establishment of options to enhance their greater provision should be prioritised at a number of neighbouring farms in an area. Where good quality semi-natural resources are available management/enhancement of these existing habitats rather than the creation of new ones may be the better conservation strategy since these habitats offer a range of complementary resources for wildlife in farmland landscapes
- The success of any ELS agreement relies heavily on the quality of plot establishment and maintenance. It is clear that options vary in this respect (see Table 2.1 below). Often if a sown option is not doing well after the first year, it will be unlikely to improve and should be abandoned/ re-established. As with conventional crops, ground conditions at time of establishment and subsequent weather conditions can contribute to the longer term quality of a plot.

Treatment	Option Code	Description	Establishment	Maintenance
Cross		Hedges cut annually post-harvest	1	1
Compliance		Protective buffer zones (hedgerows & water)	1	1
Entry Level	EB1	Relaxed hedgerow cutting (once every 2 yrs)	1	2
Stewardship	EE3	6m sown grass margins (4 grasses)	2	1
	EF2 b	Biennial wild bird seed mixture	2	2
Entry Level	EB1	Relaxed hedgerow (cutting once every 2 yrs)	1	2
Stewardship	EE3 (+)	6m sown grass margins (5 grasses & 6 forbs)	2	1
Extra (ELSX)	EF1	Wildflower corners (4 grasses & 25 forbs)	3	1
	EF2a;	a) annual wild bird mix	2	3
	EF2b;	b) Biennial wild bird seed mixture	2	2
	EF2c	c) 'Bumble-bird' mix	2	3
	EF4	Nectar flower mixture (4 legumes)	3	3
	EF8	Skylark plots: 20 plots per 10ha field	2	1
	EF11	Annually cultivated margins (uncropped)	1	3

Table 2.1. Summary table of options sown across Hillesden and qualitative assessment ease of their establishment and maintenance. 1=straightforward; 2=moderate (e.g. some ground preparation, weed control, rotation); 3=more involved (e.g. careful ground prep., cutting regimes, re-establishment). Overall scores (and mean score per option): CC=2 ($\bar{x}$ =1.0); ELS=8 ($\bar{x}$ =2.7); ELSX=37 ($\bar{x}$ =4.1)

Advice on specific margins

- *EF4 nectar flower mixtures* should not include grasses (reducing competition and aiding re-establishment). Inclusion of important forage like Red clover (*Trifolium pratense*) and Alsike clover (*T. hybridum*) as major elements of the mixtures means they are unlikely to last for the 5 years of an ELS agreement due to their short-lived nature. If possible patches should be relocated within the landscape after 2-3 years possibly in rotation with wild bird patches. Consideration should be given to including early and late season flowering species to provide resources for early emerging queens and males and queens later in the season.
- *Bird patches*.
    - To maximise cost-effectiveness, patches need to be managed to be as productive as possible. Increasing seed availability will increase bird numbers in

winter (at least in terms of the range of numbers present at Hillesden); thus increases in both patch quality and patch size should be beneficial.

  - Bigger, more productive patches may help meet the "hungry gap", i.e. supplies will last longer and may also help conserve "natural" food supplies for longer into the winter.
  - Supplementary feeding may also address the "hungry gap", (not tested directly at Hillesden) but would require a range of seed sizes to be supplied to cater for more than just the large-billed buntings and finches.

- *Hedges* managed under the ELS year option EB1 (Natural England, 2010) are unlikely to achieve the aim of substantially increasing biomass of berries and provide relatively little benefit to wildlife as food resources for over winter compared to typical annual cutting. Other studies suggest that later biennial cutting, to allow berries to be utilised through the winter, is preferable but on heavy land this would necessitate of wide grass margins to allow machine access.

3 Suggested areas of future work

- Quantify the effects of ELS options/ agreements on population demography and population dynamics of a range of taxa across a landscape gradient.
- How does land use context and land use change the relative impact of ELS?
- Is the response of species to ELS behavioural (i.e. are they acting as population sinks) *vs.* population level (i.e. promoting them as sources)?
- Identify the synergies and antagonisms between conservation of different taxa/functional groups and ecosystem service in response to different options/ agreements.
- Understand how to scale the observations of population trends at local scales to predict national effects?

4 Background

Farmland biodiversity in the UK, and throughout Europe, has declined substantially during the last few decades of the 20th century, largely attributable to agricultural intensification (Krebs *et al*., 1999; Donald *et al*., 2001; Marshall *et al*., 2003; Thomas *et al*., 2004). Despite a general lack of strategic and widespread monitoring programmes changes for many taxa are relatively well documented and accepted by policy makers. The declines have occurred particularly among those species most closely associated with cropland including the arable weed flora (Preston *et al.* 2002; Robinson & Sutherland 2002a; Heard 2003), non-pest invertebrates (Aebischer 1991; Robinson & Sutherland 2002a), insect pollinators (Biesjmeiet, Williams and Osborne) and farmland birds (Fuller *et al.* 1995; Siriwardena *et al.* 1998; Chamberlain *et al.* 2000; Newton 2004).

In general reconciling the conservation of biodiversity with increased agricultural production has involved two strategies, 'Land sparing' and 'Land sharing' (Green *et al.* 2005). In practice these two approaches represent endpoints on a continuum of approaches to managing biodiversity and the scale with which either is applied is contingent on the characteristics of the taxa/habitat of interest. Land sparing is considered more applicable to the conservation of species associated with pristine natural habitats and requires large, contiguous areas to be protected for wildlife conservation while the intervening land is farmed intensively for maximum production. In contrast, land sharing is consistent with a more 'wildlife-friendly' farming approach and integrates conservation with more extensive farming practices. In practice these two strategies form a continuum of interventions. Despite a lack of supporting evidence there has been a strong policy drive for the latter strategy in Europe, largely through the agri-environmental measures incorporated into the Common Agricultural Policy (CAP). This led to the development of agri-environment schemes (AES) that aim to restore and enhancing semi-natural habitats in farmland to counter declines in biodiversity.

AES were first introduced in the EU in 1992 and have been compulsory since 2003. They have comprised a major component of UK Government policy for almost two decades with the overall aim of reducing the loss of farmland biodiversity. In general these policies have been aimed at improving habitats across the whole of the agricultural landscape to affect populations at the national scale. The AES have relied heavily on the establishment

and management of field margins as a means of integrating both agronomic and environmental objectives on farmland. These managed margins are potentially effective for enhancing biodiversity and are considered as important conservation measures. However despite their popularity, there have been relatively few attempts to assess their relative value for farmland biodiversity either singly or in combination with each other at a landscape scale. However what evidence there is suggests both local and regional variability in delivery of environmental benefits on farmland. For example, in the UK positive effects on farmland birds have been restricted to targeted species in specifically designed schemes in localised areas rather than national, population level increases (Perkins, Field 2011). This is also reflected in evidence collected from across the EU and has led to a general questioning of the benefits of AES for conservation (Kleijn and Sutherland 2003).

The variable delivery of environmental benefits on farmland has led to the development of hypotheses that try to reconcile the impacts of land use and landscape context on conservation interventions (Kleijn *et al* 2011). The first suggests that conservation will be most effective in more extensively farmed landscapes because the potential biodiversity increase per land use intensity change is highest. This results from greater landscape heterogeneity in more extensive landscapes providing more niches (and less competition) for species to coexist. The second predicts that conservation will be most effective in simple landscapes (2-20% semi natural cover as opposed to 'cleared' landscapes with <2% semi-natural habitat) because biodiversity in managed fields is constantly subsidised by colonisation from surrounding species-rich land. There is increasing evidence for the second hypothesis (Heard *et al* 2007, Carvell 2011). However it is unlikely that either hypothesis can currently explain AES effectiveness since the results of any AES scheme/ conservation intervention will be contingent on appropriate targeting of taxa, appropriate measurement of response and the magnitude of the contrast between the farmed habitat and the intervention.

Pilot studies conducted prior to the introduction of the latest English AES (the two tiered "Environmental stewardship scheme"; Natural England 2010) suggested that uptake of the basic level (Entry Level Scheme, ELS) would be high with over 80% of farmers joining. However it was unclear how effective the scheme would be for biodiversity enhancement and a number of issues were highlighted: (i) farmers tended to favour a small proportion of

the (then) 55 available (predominantly margin) options; and (ii) many farmers tended not to apply those options that would provide most environmental benefit in their particular circumstances. Since its inception ELS has grown to cover 5.6 million ha (60% of the utilisable farmland) and has an annual budget of €202 million. It comprises over 60 management prescriptions either to enhance or to create wildlife habitat on farmland. Most of these have broad environmental aims and are simple and cheap to implement. However overall there has been relatively poor uptake of the within –crop and more interventionist options that are likely to be more effective at delivering environmental benefits (Boatman *et al* 2008).

The Hillesden experiment was conceived at the beginning of the ELS scheme to compare the effects on biodiversity of conventional intensive arable farming under cross compliance with: (i) predicted typical ELS option uptake; and (ii) enhanced and targeted ELS option uptake. Rather than take the traditional small plot experimental approach the experimental design applied combinations of margin options to parcels of land it sought to gain a more holistic view of the impacts on wildlife at this landscape scale. This had the added benefit of allowing monitoring of more mobile species e.g. bumblebees, mammals and birds that utilise larger areas and are amongst the most threatened and declining taxa associated with farmed landscapes.

4.1 **Objectives**

This study undertook a quantitative evaluation of the cost-effectiveness of ELS in conserving and enhancing biodiversity at the farm scale. The following specific objectives were addressed:

i. Establish a farm-scale randomised block experiment to compare the effects on biodiversity of conventional intensive arable farming under cross compliance with: (i) typical ELS option uptake; and (ii) enhanced and targeted ELS option uptake.
ii. Monitor the effects of these treatments on (i) habitat quality and food resources, and (ii) the abundance, diversity and population dynamics of key farmland taxa over a 5 year period;
iii. Use the experiment to promote knowledge transfer between researchers and practitioners (farmers, advisors, scientists, scheme administrators) through training days and workshops.

5 Summary of methods

5.1 **Study Site**

This study was undertaken at the Hillesden Estate, a *c.* 1000 ha arable farm in Buckinghamshire, England (Long. 1˚00'01"W; Lat. 51˚57'16"N). The farm is situated on heavy clay soils and is entirely arable with a simple rotation of autumn-sown winter wheat, oilseed rape and field beans cropped in large blocks. The farm has had little history of environmental enhancement associated with agri-environment schemes prior to the experiment. The landscape around the farm is predominantly arable in land cover (51%), followed by grass (31%), woodland and woody shrubs (8%) and only 2% urban (small villages). Mean crop yields on the farm during the period 2006-2010 were: first winter wheat = 9.1 t ha^{-1}; second wheat =7 t ha^{-1}; winter oilseed rape = 3.3 t ha^{-1} and beans = 3.1 t ha^{-1}. These values are favourable when compared to the average yields in the SE region over the same period (winter wheat = 7.9 t ha^{-1} and oilseed rape =3.3 t ha^{-1};Defra statistics) and confirm the farm as a productive and fairly intensive arable unit.

5.2 **Experimental Design**

In 2005 we established a farm-scale randomised block experiment. The farm was divided into five replicated experimental blocks of c. 180 ha, (variation in block area reflects the delimitation of blocks by existing field boundaries) with no buffer gaps between the blocks. Areas of semi-natural habitat such as hedgerows, field margins and trees were evenly distributed across the farm. Each block was separated into three areas of c. 60 ha and each of these was randomly allocated one of three treatments (for further details see Table 5.1 and Appendix 12.1 for experimental layout):

i. **Cross Compliance (CC)**: annual post-harvest hedge cutting and buffer zones to protect hedges and water courses. This simulated a 'business as usual' scenario of farming and acts as the control treatment.

ii. **Entry-Level Scheme (ELS):** 1% of land removed from production to achieve 30 points ha^{-1}, with option preference based on the average ELS uptake (as predicted in 2005). A small number of simple options were established, namely hedges cut every two years, one winter bird food patch and some grass margins.

iii. **Entry-Level Scheme Extra (ELSX):** 5% of land removed from production to achieve 45-50 points ha^{-1}. This included a more diverse range of options including biennial

hedge cutting, three sorts of bird food patch and a range of grass and flower rich margins and patches.

Treatment	Option Code	Description
Cross Compliance (CC) **Control treatment**		Continuation of conventional intensive arable management subject to the rules of Cross Compliance
		Hedges cut annually post-harvest
		Introduction of protective zones to buffer hedgerows and watercourses
Entry Level Stewardship(ELS) **(1% of land removed from production)**	EB1	Relaxed hedgerow cutting once every 2 years
	EE3	6m grass margins by sown with a mix of 4 grasses @ 20kg ha-1 aimed at creating new habitat for small mammals, invertebrates and birds
	EF2 b	Wild bird seed mixture patch (0.25ha) sown with biennial seed mix
Entry Level Stewardship Extra (ELSX) **(5% of land removed from production)**	EB1	Relaxed hedgerow cutting once every 2 years
	EE3 (+)	6m grass margins sown with 5 grasses & 6 good performing forbs in a 95:5% mix @ 20kg ha-1 aimed at creating new habitat for small mammals, invertebrates and birds and provide nectar and pollen resources
	EF1	Field corners sown with 4 grasses & 25 forbs in a 90:10% mix @20kg ha-1
	EF2a; EF2b; EF2c	Wild bird seed mixes: 3 patches (0.5ha); 3 different seed mixtures: a) annual 'deluxe' mix, b) biennial 'ELS tall' mix, c) 'Bumble-bird' augmented to include plant species that provide nectar resources for pollinators as well as seed resources for birds
	EF4	Nectar flower mixture sown with 4 legumes @15kg ha-1 aimed at providing food resources for nectar feeding insects e.g. bees and butterflies
	EF8	Skylark plots: 20 plots in 10ha Winter Wheat field aimed at providing space for ground nesting skylarks
	EF11	Annually cultivated margins (uncropped) to encourage rare plants and foraging sites for seed-eating birds

Table 5.1 Details of the options contained in each treatment. For further details of the species sown for each margin type see appendix 2. Note actual areas sown varied slightly around given areas.

5.3 **Summary of monitoring**

Monitoring at Hillesden sought to quantify the effects of farm-scale application of the three main treatments on (i) habitat quality and food resources, ii) abundance & diversity of a range of taxa and (ii) the population dynamics of key taxa (Table 5.2). The selected key taxa

were mobile species which were either a) likely to respond to environmental enhancement at the farm-scale; or b) were the key resource requirement for these species (e.g. insect abundance in the summer for farmland birds). These included indicator species (e.g. farmland birds), umbrella species (e.g. butterflies), and key species important for ecosystem function (e.g. pollinators). Where possible, monitoring focused on the key factors influencing population dynamics, such as breeding success and overwinter mortality.

Taxa monitored	Timing	Frequency yr^{-1}	Methods
PLANTS: Hedge berry production	Sept- Oct	1	• 20 × quadrats (0.5m × 0.5 m); 1.5m above the ground. • All hard fruited counted, picked and weighed. • Counts of soft fruits
PLANTS: Seed production in bird plots	Sept- Oct	1	• 5 × quadrats (minimum 1 m × 0.5 m) • Counts of individual plants (seeding/non-seeding). • Seeds collected, counted, dried and weighed.
PLANTS: Dicotyledon flower abundance	May-Aug	4	• Estimates of flower abundance in 2 quadrats (2m×2m).
PLANTS: Phytometers	May/June	1	• Estimates of seed production in species sown centrally in each treatment.
INVERTEBRATES: General invertebrate abundance			• Vortis suction samples taken At 5 points (> 5m apart) along the centre of different margin types. • Samples sorted to key groups and taxa • Counted & biomass obtained after drying.
INVERTEBRATES: Pollinator abundance – pan traps	June	1	• Water filled traps, passive sampling • Catch sorted to broad taxonomic groups
INVERTEBRATES: Pollinator abundance - Transects	May-Aug	4	• Foraging bumblebees and butterflies were counted along a fixed 2m × 50m transects in the centre of each treatment patch
INVERTEBRATES: Pollinator abundance - timed observations	May-Aug	4	• 2 quadrats (2m×2m) observed for 2 minutes on each occasion • All insects measuring ≥3mm & flower 'units'
INVERTEBRATES: Moth trapping	May/July	2	• Portable light traps placed in the centre of each treatment for a night
INVERTEBRATES: Trap nests	Spring-Autumn	1	• Pairs of traps reeds & bamboo canes next to boundaries at centre of each treatment unit. • All emerging individuals were identified to species.
INVERTEBRATES: Bumblebee colonies	Summer		• Commercial *Bombus terrestris* colonies in each treatment in year 2
VERTEBRATES: Breeding bird census	Apr-June	4	• Bird activity in hedges & 10m either side were mapped monthly
VERTEBRATES: Breeding bird productivity	June-Aug	12	• Bird breeding activity and juveniles 10m either side were mapped weekly
VERTEBRATES: Winter birds	Nov-Dec	3	• Bird activity in hedges & 10m either side were mapped monthly
VERTEBRATES: Nest boxes	Apr-July	16	• 5 tit boxes, 1 sparrow box per treatment • 7 Breeding success parameters recorded.
VERTEBRATES: Small mammal trapping	Nov-Dec & May	2	• Longworth trapping in spring and autumn (Years 0, 1, 3 and 5). • Two central grass margins per treatment • Caught individuals were marked using pit tags placed subcutaneously.

Table 5.2. Summary of taxa monitored across 5 years (2005/6-2010/11), frequency of monitoring and brief method description.

5.4 **GIS**

In August 2007 remote sensed data were acquired for the whole site (ca. 25km2) using the Specim AISA Eagle (400–970 nm) and Hawk (970–2450 nm) hyperspectral sensors. These data, in conjunction with detailed ground survey information, were used to construct a high-resolution 3-D structural and land-cover habitat map with a pixel size of 0.5m×0.5 m. The images were geo-referenced and classified using ERDAS Imagine (ERDAS Inc., Georgia, USA) software, while subsequent analysis was performed in ArcGIS 9.2 (ESRI Inc., California, USA). For a number of radii surrounding each taxa specific sampling point (see

Figure 5.1 and methods below), individual land parcels were identified from the land cover map as: (1) Winter Wheat; (2) Oilseed Rape; (3) Permanent Grass Cover (excluding field margins); (4) Hedgerow area; (5) Tussock Grass margins, i.e. those containing only grass species; (6) Floristically Diverse field margins, i.e. those containing a forb component such as the Pollen & Nectar margins, Wild Bird margins and Wildflower margins 7) Woody plant (a composite of the volume of shrubs, hedgerows and trees).. Finally, the overall diversity of habitat types was determined using a derivation of the Shannon–Wiener landscape habitat diversity index (Steffan- Dewenter, 2003): *H'* Landscape = −∑pi. ln pi where pi is the proportion of the landscape that was comprised of each of the habitat *i* elements described above.

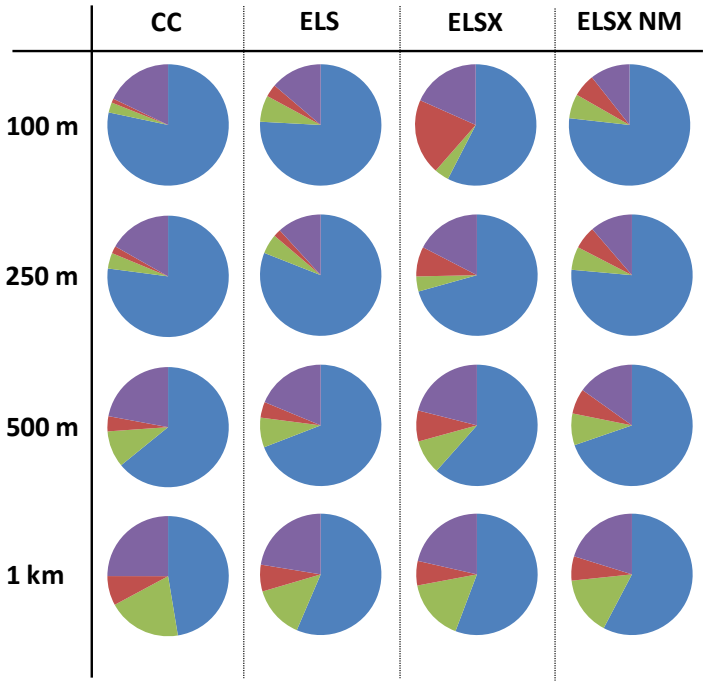


Figure 5.1. Example of % cover of different habitat types at different spatial scales (100m, 250m, 500m and 1km) around moth traps in different treatment types (CC, ELS, ELSX, and ELSX NM) determined from land use map: arable crop (blue);

grassy habitat (green); forb-rich habitat (red); and woody plant cover (purple). Note similar information was generated around all taxa specific sampling points.

6 Habitat Quality & Food resources

6.1 Winter food resources for birds

While seed production is a fundamentally important process for plant population dynamics it also provides a major source of food for a diverse range of animals. Within the farmed landscape these typically include granivorous mammals, birds and insects (e.g. ants and beetles), but can also include a wider taxonomic range of species like slugs and earwigs. The dynamics of seed resources in agricultural landscapes is poorly understood and has been identified as a key area for further research (Butler *et al*., 2010).

Many seed-eating farmland bird species rely heavily on weed seeds for food, particularly outside the breeding season (Hole *et al* 2002; Moorcroft *et al* 2002; Robinson & Sutherland 2002). The declines of these birds have, in part, been attributed to changes in over-winter adult survival (Siriwardena *et al.* 1998b, 2000; Newton 2004) linked to reductions in winter food supplies. Birds may simply die later in the winter when food resources run out. Declines in arable weed abundance since the 1940s (Robinson & Sutherland 2002) and increases in winter cropping (and concomitant decrease in over wintered stubbles) have reduced stocks of winter seed known to provide food and thus underpin species of conservation concern. Such negative effects may be offset by either increasing the abundance of weed seeds e.g. through lowered weed control thresholds or by providing additional seed resources elsewhere in the farmed environment e.g. through AES prescriptions.

Many of the AES scheme options targeted at farmland birds aim to create appropriate foraging, breeding or overwintering habitat. Some are designed to specifically provide food resources. In the UK, for example, these include the wild bird cover option in which plants are cultivated to produce copious amounts of seed over the winter and the Conservation Headlands in which invertebrate-rich cover is generated at the edge of cereal fields. However relatively few studies have actually quantified seed food resources within different margin treatments. Here we present data from a range of key habitats across a whole farm. Understanding which habitats provide the highest seed abundance, biomass and energy is crucial for finding ways of optimising and restoring seed-feeding species on

degraded farmland. We quantified the abundance and diversity of pre-dispersal seeds in non-crop, sown habitat. We also quantified pre-dispersal berries within hedgerows under differing management options, because they are known to be important sources of food for farmland animals (Snow and Snow, 1988).

6.1.1 Objectives

We had three overall objectives: i) to assess the effectiveness of establishing a range of ELS options aimed at providing seed resources for birds; ii) to estimate the abundance, biomass and energy content of seeds and fruits provided by these options at both the option and whole treatment scale; iii) to link the abundance of seeds to the predicted and actual abundance of farmland birds and mammals.

6.1.2 Methods

We evaluated the seed and berry production of a selection of different ELS options using the following methods.

i) *Hedge berry production.* A 200 m section of internal hedgerow was selected in each treatment replicate. Berry biomass production was estimated at its peak in September/ October each year from twenty equally spaced 0.5m × 0.5 m quadrats (ten each side of the hedge) at 1.5m above the ground. For each quadrat, all hard fruited berries (e.g. Hawthorn, Blackthorn, Rose hips) were counted, and from 2007 picked, dried at 80°C for 24 hours and weighed. The numbers of soft fruits (Blackberry, Elder heads, Bryony, Privet) per quadrat were counted.

ii) *Seed production in bird food plots*. This was assessed at peak abundance in September/October each year. For each plot, five quadrats (minimum 1 m × 0.5 m per qaudrat, minimum total area = 2.5m^2) were sampled at random (avoiding the first 3 m from the edge, but utilising the whole plot). The number of individuals (ramets) rooted in the quadrat were counted and recorded, along with their reproductive status (seeding/non-seeding). Seeds and seed heads from each of the seeding species were collected. Seeds were separated from the chaff, counted, dried at 80^{o}C for 24 hours and then weighed.

We quantified the numeric abundance, biomass and energy provision of sown and unsown seeds per m^2. The energy content of each seed species (kJ/g) was obtained using a database constructed by the Royal Society for the Protection of Birds (see Gibbons *et al*., 2006). For some seeds it was difficult to find species specific data and in these cases we

calculated a surrogate weight and energy content as the mean of the other species in its genus. We were unable to quantify the energy values for hedgerow berries because of a paucity of calorific data.

We estimated the number of granivore days supported by each treatment. First we calculated the daily energy requirement of an average individual, free-living granivorous farmland bird using the allometric relationship between field metabolic rates (total energy cost an animal pays during a day) of passerines and body size (Nagy 1987). We did this by calculating the daily energetic requirement for the following seven mainly granivorous Farmland Bird Index (FBI) species: Corn bunting, Goldfinch, Greenfinch, Linnet, Reed bunting, Tree sparrow, Yellowhammer, and took the geometric mean value of energy requirements per day. We then estimated the total energy provided by seeds in each treatment by multiplying the total area of each option in each treatment replicate and its mean energy yield per m^2. In turn this value was divided by the mean daily energetic requirement for an average passerine to give a total number of granivore days per treatment. We used Repeated measures ANOVA on transformed data to analyse across years and Generalized Linear Models with appropriate error structures and link functions to investigate the effects of margin type and treatment on seed biomass, energy provision and number of granivore days.

iii) *Seed depletion* in bird plots was assessed using a simple scoring system to assess the availability of seed through the winter of 2007/8. Monthly visits (October – March) were made to all patches and the % of seed remaining on individuals was estimated for sown species at random locations throughout the plot. Mean values per species and per treatment were calculated.

6.1.3 Results

6.1.3.1 Hedge Berry Production

The hedges were fairly typical mixed hedges which in terms of berry counts were >90% dominated by three species (*Crateagus monogyna,* 62%; *Rubus fruticosus agg*., 20%; *Prunus spinosa* , 10%). Other species at relatively low densities included *Rosa* spp. (4%), *Tamus communis* (2%) with *Cornus sanguine, Ligustrum vulgare, Solanum dulcamara, Bryonia dioica, Malus spp, Crataegus laviegata, Symphoricarpos spp.* and *Sambucus nigra* (heads) all producing less that 1% of total berries. Total counts of soft and hard fruits from 2006-2010

showed significant year effects (repeated measures ANOVA, square root transformed count data; total fruit $F_{4,48}$=7.64, P<0.001; soft fruit $F_{4,48}$=13.87, P<0.001; hard fruit $F_{4,48}$=10.43, P<0.001; proportion soft berries ($F_{4,48}$=16.02, P<0.001). There were no significant difference in total numbers between treatments (Figure 6.1) or interactions between treatments and time although total counts were markedly higher in the CC treatment in 2009 and when analysed for this year alone this effect was significant ($F_{2,8}$=4.77, P=0.043). Total weight of hard berries again showed significant year effects (Repeated Measures ANOVA, log 10, $F_{3,36}$ =11.66 P=<0.001) but no significant effect of treatment, although in general the CC treatment appeared to have higher yields. Overall yields varied from a low in 2008 of 10.38 g $5m^{-2}$ (±se =8.12-13.25) to a high in 2009 of 68.39 g $5m^{-2}$ (±se =53.55-87.33).

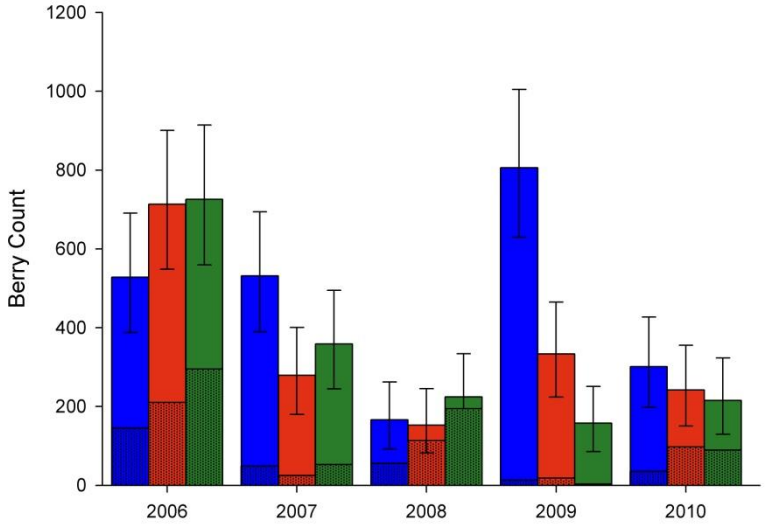


Figure 6.1. Total (whole bar), soft (hatched bar) and hard (unhatched bar) berry counts (values= back transformed) per $5m^2$ in each treatment (red =CC, blue =ELS, green =ELSX) across 5 years. Error bars are 1SE of total count.

6.1.3.2 Seed production in bird plots.

There was a significant effect of treatment (Repeated Measures ANOVA, log_{10} transformed, $F_{1,4}$=*61.38*, P=0.001) and year (Repeated Measures ANOVA, log_{10} transformed,$F_{4,32}$=*24.56*, P<0.001) on total seed production. The effect of year was dominated by pulsed seed production that meant that produced greater yields every other year. Overall seed yields in the ELSX treatment provided approximately 8 times more seeds for farmland birds in winter compared with ELS.

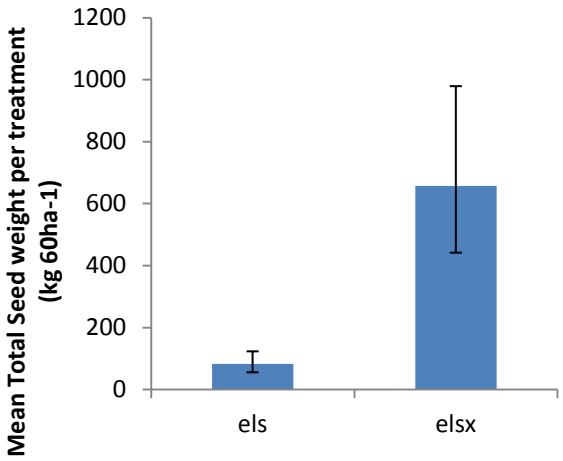


Figure 6.2, Mean seed yield (all species combined) per treatment across all years 2006-2010. Error bars are 1se.

As expected similar effects on energy provision were observed with significant interactions between effects of treatment ($F_{1,4}$=*18.4*, *P*=0.013) and time ($F_{4,32}$=*9.76*, *P*=0.004) which showed biennial fluctuations. On average the ELSX treatment yielded over 13 times as much energy when compared to the ELS treatment (9.07 x 10^6kJ in the ELSX treatment and 6.9 x 10^5kJ in the ELS treatment). There was a significant interaction between treatment and time in the potential number of granivore bird days provided by the treatments (Figure 6.3). However the granivore predicted days were not significantly correlated with the winter granivore counts per km (see Bird section below for method details) in 2006 and 2009 (log 10 values, *F*=0.10 *P*= 0.758).

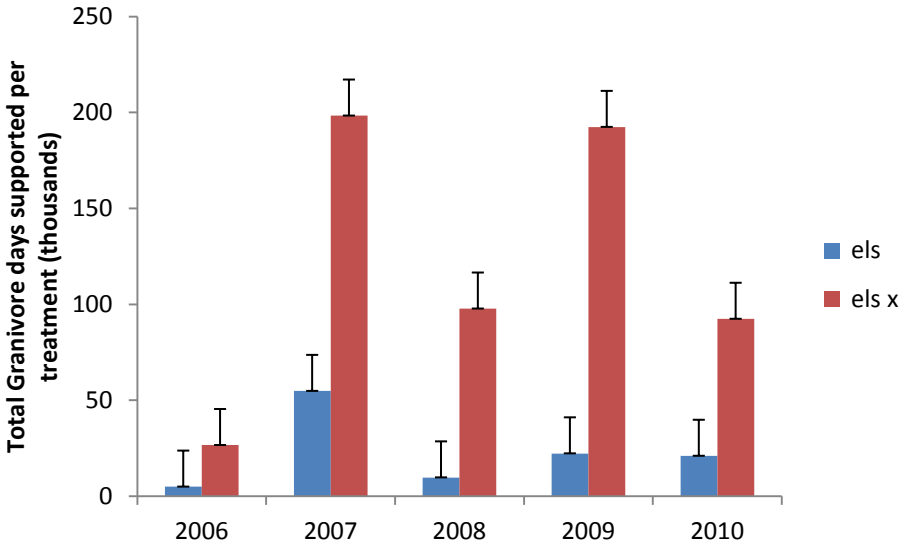
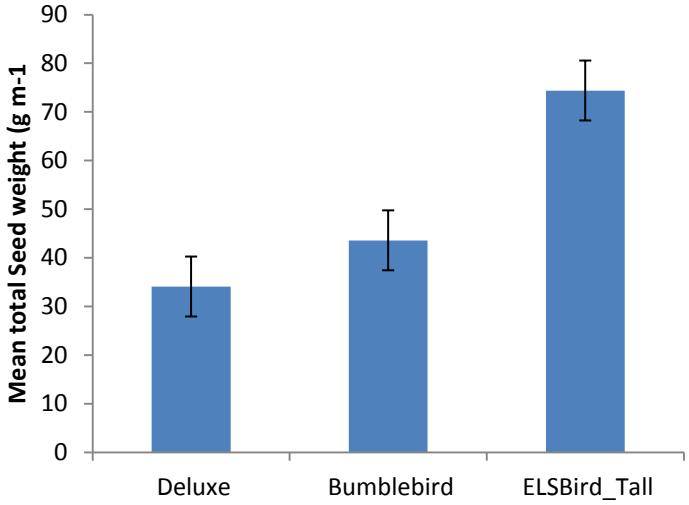


Figure 6.3. Number of granivorous passerine days supported per treatment (60ha) by sown bird food plots

Mixtures varied widely in terms of seed provision. Total seed biomass (log 10 transformed) was significantly higher ($F_{2,8}$= 9.86, *P*=0.007) with the EF2b (ELS Bird Tall) mix out yielding the other mixes by a factor of just over two. This was also reflected in the energy provided by the mixtures ($F_{2,8}$= 5.02, *P*=0.039, $\log_{10}$ values), 1.6-2.4 times higher. In terms of energy provided the ELS Bird. Analysis of the variability of mean seed production (log 10 coefficient of variation) showed that the bumblebird option was significantly less variable in seed delivery through time compared to the other mixes ($F_{2,8}$ =3.77, *P*=0.07).

a)



b)

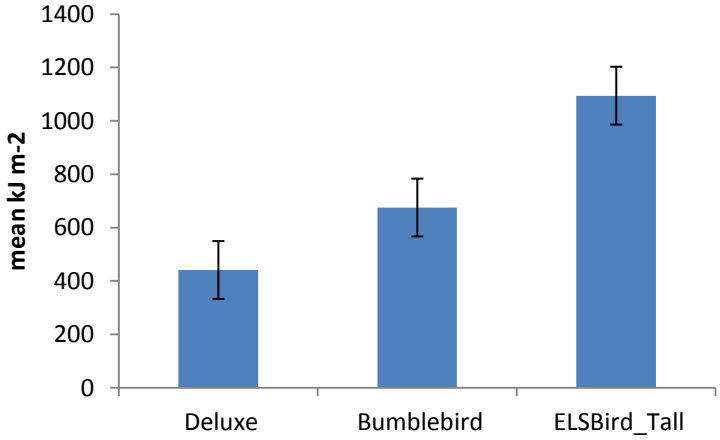
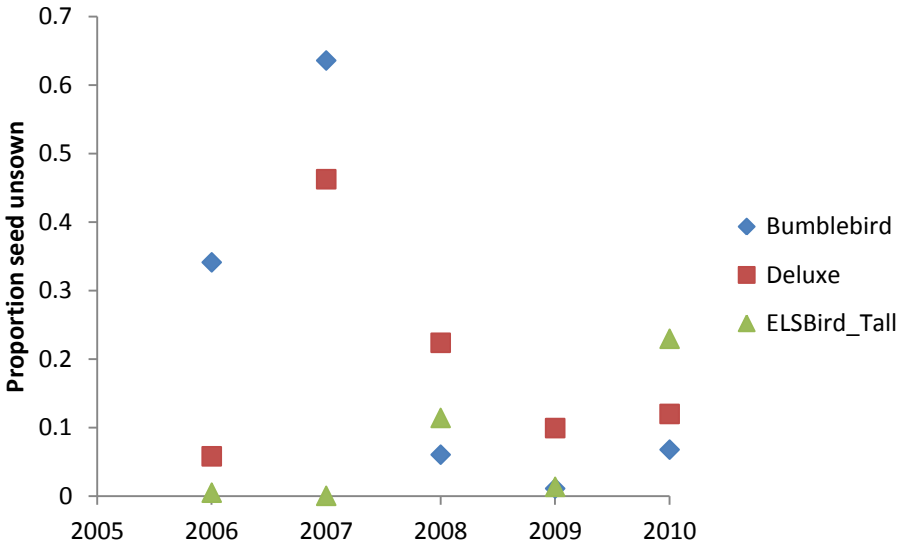


Figure 6.4a) Mean total seed yield (g m^{-2}) and, b) mean total energy content (kJm^{-2}) of sown seed in different bird plot options 2006-2010. Untransformed means are presented for clarity. Error bars are 1se.

The weediness (proportion of seed weight that was unsown) of the three types of patches showed a significant interaction with year ($F_{8,47}$= 3.83, *P*=0.022, arcsine transformed values). In bumblebird and deluxe options weediness decreased through time while in ELS Bird tall it

increased, by the last year of the experiment the values had converged somewhat to between 7-23%. Overall and averaged over all years, the ELS Bird Tall option was less than half as weedy (7%) as the other options.



6.1.3.3 Seed depletion 2007/8

In general more than 50% of the seed produced had gone by late November with more than 90% gone by January. There were differences in seed shedding rate: Kale & Millet shed> Fodder radish & Triticale > Fodder beet.

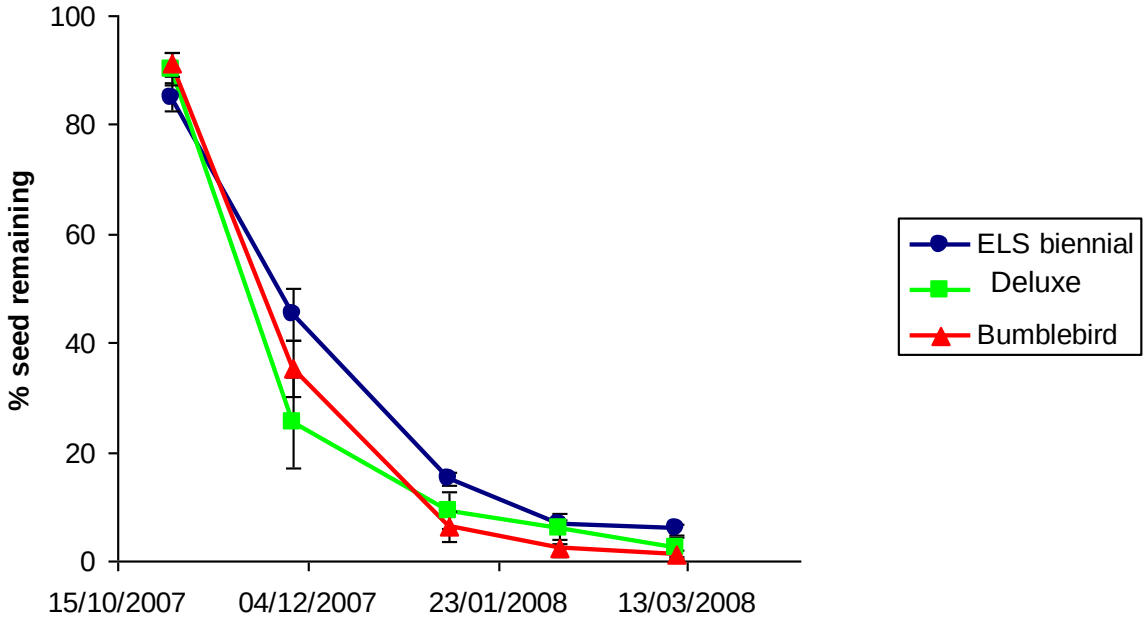


Figure 6.5 Mean % seed remaining across all sown species in each bird food option though the winter of 2007/8. Error bars are 1 s.e.

6.1.4 Discussion

6.1.4.1 Hedge Berry production

This study evaluated the effect of cutting cycles on hedgerow berry production over the typical length of an ELS agreement. We show that the there was no significant difference on yield and berry counts between hedges cut annually (typical non-ELS practice) compared to those cut biennially (typical ELS practice). There were large year to year differences and in one year, contrary to expectations, the annual cut hedges out yielded the biennial. The generality of this result is partly confirmed by a similar study on hawthorn (Staley *et al.,* 2011) which showed that biennially cut plots only produced significantly more flowers than those cut annually in 2 out of 5 years. Both these results contrast with those of Maudsley *et al.* (2000) who found greater berry mass on biennially cut woody species compared to those cut annually or every 3 years. However, their data was based on 1 year of sampling, in which the biennial plots had not been cut but both the annual (and 3 year) plots were cut (Maudsley *et al*., 2000). Their findings therefore related to the immediate response of woody species to the cutting regime that year.

All the cutting at Hillesden was in autumn (post harvest) as is typical of many farms on heavy land for practical reasons of access for machinery which is unable to travel when the ground is wet. Staley *et al* (2011) showed that cutting time can strongly affect berry mass when hawthorn hedges were cut biennially, with berry mass only increasing significantly on winter cut hedges plots in the second year of the cutting cycle. Timing of cutting appears to modify the growth patterns of woody species. For example, late summer cutting of hawthorn produces a greater number of shoots the following year than winter cutting and can affect physiology (e.g. leaf C:N ratios, S. Amy *pers. comm.*). In addition autumn cutting of biennial hedges depletes the berries which form on two year old growth before they can be utilised as a food by birds and mammals over winter. Hedgerow cutting regimes may also have additional conservation impacts beyond the provision of winter food resources for wildlife. For example, the decline of brown hairstreak (Thecla betulae L.) butterfly populations has been partly attributed to annual cutting of hedgerows which destroys eggs laid later in the summer (Merckx and Berwaerts, 2010).

To encourage heterogeneity in the resources provided by hedgerows across a farm ELS guidance specifies that not all hedgerows managed under each option within an ELS

agreement should be cut in the same year (Natural England, 2010). However since there is no significant difference in berry yield between the most common biennial hedgerow options (>90% of agreements) and the most common practice outside of the scheme (annual cutting) this is unlikely to occur in practice. Specific co-location of wide grass margins that enable later cutting on heavy land may be worth considering to allow berry resources to be more fully utilised through the winter. This may also allay fears that some farmers express, about the negative impacts that larger, less frequently cut (i.e. 3 year cycle) hedges may have on crop yields.

In conclusion our results suggest that hedgerows managed under the ELS year option EB1 (Natural England, 2010) are unlikely to achieve the aim of substantially increasing biomass of berries and provide relatively little benefit to wildlife as food resources for over winter.

6.1.4.2 Seed production in plots

In ELS the availability of sown seeds important in the diets of a typical granivorous bird was markedly less (approximately an eightfold difference) when compared to ELSX. In terms of mass of additional seed provided across the whole treatment (compared to CC) this equates to 0.0014 t ha^{-1} in the ELS treatment and ELSX=0.011 t ha^{-1}. At the plot level the biennial mix provided the most seeds although seed production varied (was higher every other year) due to its biennial nature which affected treatment by year interactions.

Despite variation in size and composition, the predicted energy available from seeds was similar to yield. Average energy provided by the plots was 741 kJ m^{-2}. If this value is used to calculate energy provision across the landscape i.e. (energy m^{-2} x total option area per treatment)/ 60 ha), these values equate to 11530.5 kJ ha^{-1} for ELS and 151303.4 kJ ha^{-1} for ELSX. To put this value in context we compared these predicted values with the geometric mean energy content of weed seed rain derived from three conventional break crops (beet, winter and spring oilseed rape) in the Farm Scale evaluations of GM crops (Heard *et al*. 2005). This suggests that, at the landscape scale, the ELS treatment would provide less than half the energy provided by weed seeds in stubble fields covering the same area (excluding crop seed inputs). In contrast the ELSX treatment would yield approximately four times more energy than is the landscape were seed rich stubbles.

Although we used the energy provided to predict the number of granivore days this makes the assumption that all the calorific energy is available for metabolism. However studies have shown that for various granivorous species digestive efficiency is typically from 60 to 90 % (Kendleigh 1977) and can often be even lower. Thus our predictions of granivore days supported are likely to be lower in reality. Despite the positive effects on seed availability at the treatment level, there was no clear relationship between the predicted number of granivore days and total abundance from winter bird counts. This is because birds typically forage at larger scales than the treatment areas during the winter (less so in the breeding season) and most species showed little response even at the farm scale. This could also occur because habitat use is modified by a range of biological processes, such as diet choice, influence of sward density on behaviour, predation and territoriality that means eventual impacts on birds may not be as straightforward as suggested by energetics alone. We also note that our estimates of seed provision make no assumption about when seeds are supplied through the winter. It is clear from our observations that in reality > 50% of the seed is unavailable after November with over 90% gone by January. This seriously curtails the temporal availability of resources in the later winter which is a period typically associated with the greatest food shortages for birds.

The results presented here are dependent upon the option management adopted at Hillesden. It is likely that in the wider countryside that this may not be the case and that provision of seed may be lower (H. Hicks *personal communication*). In general the variability of seed provision was lower in the annual (deluxe) mix than the other mixtures although in certain years it was not atypical for certain patches to produce about 90% less seed than the most productive one. In general weed burdens were relatively high (>10%) but this seemed to converge to between 10-23% by the end of the experiment, presumably as a consequence of better seed bed management and also the increase in the seed bank of sown species. Although weeds seeds may be of use to some bird species even if they rarely outyield sown species.

Unfortunately uptake of most of these options in ELS to date has been low (Natural England *pers. comm.*). Both Vickery *et al*. (2008) and Davey *et al.* (2010) suggest that with current uptake patterns the scheme is likely to deliver limited benefits for birds. Ensuring the quality of any introduced ELS bird patches is essential, since against a background of relatively poor uptake, their ability to fill resource gaps overwinter will be even more critical.

6.2 **Invertebrate food resources for breeding birds in the breeding season**

General reductions in invertebrate abundance along with changes in community composition have resulted from herbicide applications reducing the abundance of host plants important for invertebrates and thus invertebrate abundance (Potts 1997, Brickle *et al*. 2000; Boatman 2004; Morris *et al*. 2005). In addition non-target invertebrates may be exposed to pesticides in crops which further reduce their population abundances (Boatman 2004).

As well as population declines in birds caused by reduced overwinter survival linked to lower seed availability, a relationship between food availability during the breeding season (spring/early summer) and breeding success has been linked to population change in some bird species. This link has been suggested for other species as during this period the majority of farmland birds feed their chicks invertebrates, which provide the necessary protein for growth and the energy to resist chilling (e.g. Southwood & Cross, 2002). Reduced abundance of key invertebrate prey items reduces chick survival.

Sowing mixtures of wildflowers, seeding plants for birds or other suitable pollen and nectar sources along field margins is known to increase the local abundance of key invertebrate groups. Importantly this has rarely been linked to their availability as chick food for birds and further linked to the dynamics of actual populations of birds. Here we quantify the abundance and diversity of invertebrates in a range of options during the breeding season.

6.2.1 Objectives

We had three objectives: i) to estimate the abundance, biomass and energy content of invertebrates provided by these options at the whole treatment scale; iii) based on diet choice across bird species indentify preferred margin options.

6.2.2 Methods

We measured the abundance of a range of invertebrate groups which are important food items for farmland birds using vacuum sampling in the bird breeding season (May/June) each year. A Vortis suction sampler (Burkard, UK) was used to take samples from five points (at least 5m apart) along the centre of a wide range of margin types. Samples were sorted to key groups and taxa, counted and their biomass obtained after drying at 80^{O}C for 24 hours.

We quantified the biomass and the energy content of invertebrates within habitats at peak biomass. The mean energy content of each sample (kJ/g) was obtained from the literature using mean calorific estimates derived from bomb calorimetry of dried insect samples (Driver 1983). We converted the total mass of invertebrates into total energy content per sample. In addition we calculated the daily energy requirement of an average individual, free-living farmland bird using allometric relationships between field metabolic rates of passerines and body size (Nagy 1987). First we calculated the daily energetic value for all Farmland Bird Index (FBI) species and then took the geometric mean value. We divided the total energy available from a sampled margin by the mean bird energy requirement to give a number of bird days per margin and scaled this to the total treatment area. We also calculated a mean Relative Preference Index (RPI) based on diet data taken from Holland *et al* (2006) which detailed invertebrate taxa found in the diet of adult birds and chicks during the breeding season. We first standardised these diet data to calculate mean proportion of invertebrate taxa found in the diet of an 'average' farmland bird adult (based on a selection of 14 farmland birds: Skylark, Rook, Corn Bunting, Reed Bunting, Yellowhammer, Chaffinch, Goldfinch, Greenfinch, Yellow Wagtail, House Sparrow, Tree Sparrow, Lapwing, Stone Curlew, Quail). We then apportioned the weight of the vortis catch to these groups. RPI = $\sum p_i * w_i$, where p_i = proportion of taxa *i* in diet and w_i =mass of taxa *i* in each vortis sample. We used Repeated measures ANOVA on transformed data to analyse across years and Generalized Linear Models with appropriate error structures and link functions to investigate the effects of margin, on seed biomass and energy.

6.2.3 Results

Between 2007 and 2010 total invertebrate biomass was about four times greater across the ELS Extra and 2 times greater in the ELS landscapes compared to CC ($F_{2,8=}$20.32, P=0.001,Figure 6.6). This represents a large increase in the abundance of potential food items for fledgling chicks. The energetic values (kJ) across the treatments followed this trend ($F_{2,8=}$20.16, P=0.001) and translated in to greater numbers of average FBI adult bird days that would be supported ($F_{2,8=}$19.13, P=0.001,Figure 6.7)

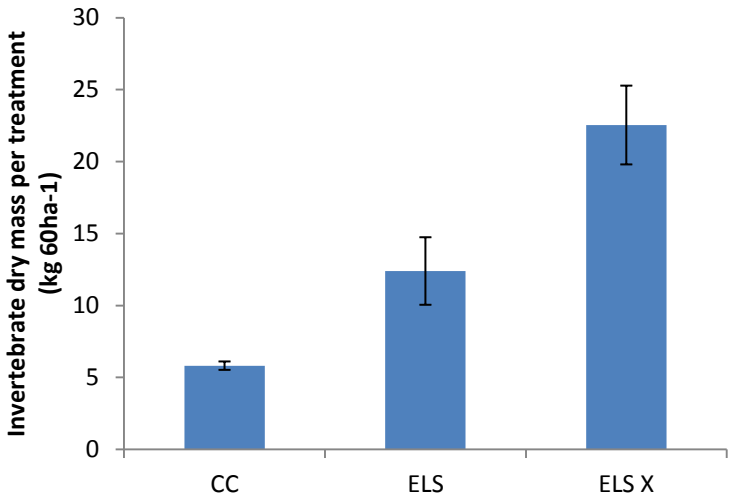


Figure 6.6 Mean total invertebrate dry weight (kg) per treatment from 2007 - 2010. .Error bars are 1 se of the mean

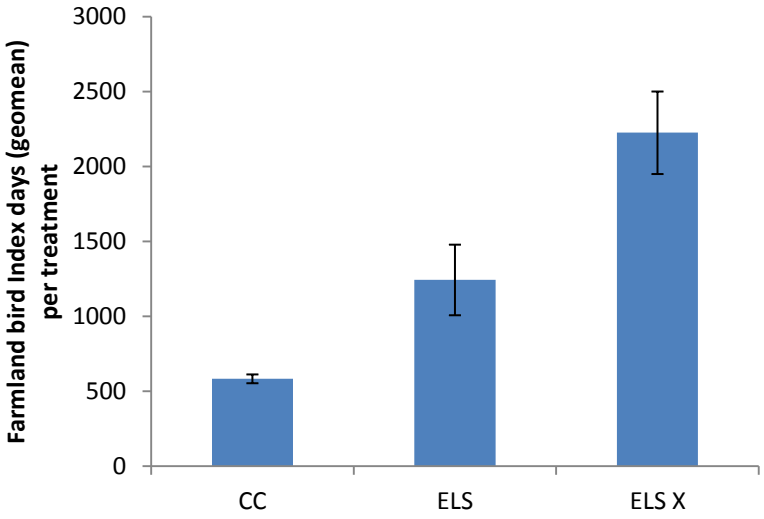


Figure 6.7. Mean total number of days of an average Farmland bird index individuals supported by invertebrates across treatments .Error bars are 1 se of the mean.

The Relative Preference Index showed significant variation across margin types ($F_{8,32}$=5.42, P<0.001). The highest scores were found for the more floristically dominated options.

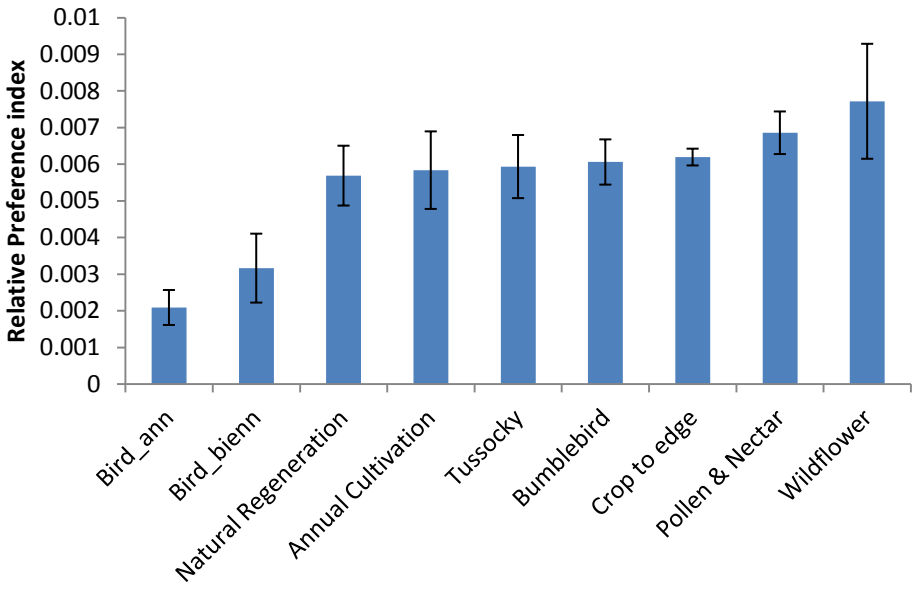


Figure 6.8. Relative Preference index (see text for calculation) of an average adult Farmland Bird Index bird calculated on average of invertebrate catches across years. Higher values indicate higher preference for prey in each habitat option. Error bars are 1 se of the mean.

6.2.4 Discussion

Although provision of spring foraging habitats are unlikely to address the main population limiting factors in birds, they are clearly critical in improving chick survival and are likely to have a positive demographic impact on a range of species. It is clear from our results that the more diverse and higher area coverage of sown options in the ELS and ELSX treatments produced a greater mass and a greater range of invertebrates across the whole landscape compared to CC. This suggests that at the farm scale, ELS scheme benefits in providing invertebrates to breeding birds will be maximised by increasing habitat heterogeneity.

Different margin options provide a range of niches for the invertebrates which provide food for farmland birds. In general the incorporation of perennial wild flowers enhances the insect fauna by providing a greater diversity of host plants for phytophagous species (Woodcock et al., 2008) as well as pollen and nectar feeders. Although caution must be exercised in interpreting the relative preference index, since it is based on diet data that may reflect availability rather than preference, it is used here for indicative purposes and is the based on the best current understanding of bird diets. The index scores show that three out of the top four options that supplied the most preferred prey items were found in ELSX options (wildflower, pollen & nectar, bumblebird) and these were all high in floral resources. Surprisingly the crop to edge also appeared to provide a large number of preferred

invertebrates. However at Hillesden this habitat is relatively similar to natural regeneration and tussocky margins being dominated by heavy land grasses and it thus was not significantly different from these options in invertebrate composition. These options have been widely promoted as important for bird food resources. In addition the availability and accessibility of prey items to foraging birds within margins will be strongly influenced by their sward structure. Although we did not investigate the way bird foraging was altered by the structure of different options, the general research consensus seems to be that patchy swards are likely to facilitate foraging by most birds. Thus, a range of different options will offer more varied foraging opportunities for a wider range of birds than the tussocky grass margins that dominate current ELS uptake figures. In general, options that promote botanical diversity will stimulate arthropod diversity along with a range of seed types for birds.

7 Abundance, diversity and dynamics of pollinators

Insects comprise the vast majority of the world's pollinators and are responsible for most crop and wildflower pollination. Approximately 70 % of the worlds 124 primary food crops are dependent to some extent on insect pollination (Klein *et al*., 2007), with this ecosystem service being estimated to be worth c. €153 billion p.a. (Gallai *et al*., 2009) and provides 9.5% of global food production value. Estimates of wild plant dependence on animal pollination vary between 78% and 94% in temperate and tropical ecosystems, respectively (Ollerton *et al.*, 2011). Proximity to natural pollinator habitat and functionally diverse wild pollinator populations enhance crop yields and there is a clear link exists between maintaining pollinator diversity and sustainable pollination services (Potts *et al*., 2010). Crop pollination by managed bees alone e.g. honeybees is often insufficient to provide a stable pollination service (Garibaldi *et al.* 2011).

However, declines in many insect pollinators have been observed worldwide (Potts *et al*., 2010) and could have very serious ecological consequences. In the UK evidence for declines in the abundance and diversity of UK wild pollinator populations, most notably of bees, butterflies and hoverflies, have been inferred from incidental records at the 10 km scale accumulated over long periods and through a range of sampling methods Across Europe evidence suggests widespread declines in both the diversity and abundance of many

wild and managed pollinators with observed, landscape-scale declines in bee and hoverfly (Biesmeijer *et al.*, 2006) diversity. Extinction, reduced abundance and range contractions of butterfly and bumblebee species have also been reported. In addition while there has been a global increase in managed honey bees Europe has experienced extensive declines in wild, feral and managed honey bees over several decades.

Declines in insect pollinators are attributed to a variety of factors, including habitat loss, pesticides, pathogens and climate change. Agricultural intensification and land-use change have been major drivers of biodiversity loss over recent decades (Tilman *et al.* 2001) and resulted in the loss and fragmentation of natural and semi-natural habitats (Ewers & Didham 2006; Winfree *et al.* 2009; Bommarco *et al.* 2010; Öckinger *et al.* 2010). These changes in landscape occur at multiple different spatial scales (Tscharntke *et al.* 2005). At the landscape scale, increases in field size and reductions in mixed farming have resulted in simplified landscapes with little non-crop area. At the local scale, intensification of resource use and increasing inputs have led to the simplification of remaining semi-natural habitats with knock-on effects for many taxa (Robinson & Sutherland 2002b). The response of pollinators to landscape structure can be complex, with species functional traits (such as diet breadth, sociality and dispersal ability) modulating these responses (Thomas 2000; Bommarco *et al.* 2010; Öckinger *et al.* 2010; Williams *et al.* 2010). For example, small-bodied, generalist bees may be more sensitive to habitat loss than large-bodied species (Bommarco *et al.* 2010). Understanding how the structure of pollinator communities respond to anthropogenic induced environmental change has considerable implications for their conservation and management of the ecosystem services they provide (Steffan-Dewenter *et al.* 2002; Kremen *et al.* 2007; Ricketts *et al.* 2008; Keitt 2009).

ELS includes specific options targeted at pollinators, aiming to enhance the supply of pollen and nectar sources through the sowing of conservation flower mixtures at field margins (Natural England 2010). This has been shown to significantly enhance the local density and diversity of key pollinators on arable land when compared with conventional cropping or other less targeted ELS options (Pywell *et al.* 2006; Carvell *et al.* 2007). How much of this high quality foraging habitat is needed and how it should be distributed within agricultural landscapes is not yet known, and is a key question to address if the impact of such schemes is to be maximised across different regions (Lonsdorf *et al.* 2009). It is,

however, recognised that both foraging behaviour and population size of pollinators are likely to be determined by patterns of resource availability at landscape, rather than local, scales (Knight *et al.* 2005; Osborne *et al.* 2008).

An increasing number of studies have established correlations between species richness and density of flower-visiting bees on focal plots and landscape context (Steffan-Dewenter *et al.* 2002; Westphal *et al.* 2003). It is also clear that farming systems can interact with landscape context to determine local bee diversity, such that, for example, organic farming has a stronger positive effect on bee diversity in homogeneous landscapes (Holzschuh *et al.* 2007; Rundlöf *et al.* 2008). However, while it has been proposed that conservation measures promoting the sowing of flower mixtures should focus on more intensively farmed (though not highly simplified) landscapes (Tscharntke *et al.* 2005; Isaacs *et al.* 2009), there is no experimental evidence for this. Importantly, any effects of sown flower mixtures need to be assessed relative to existing resources supplied by natural vegetation in conventionally managed control areas (Kleijn *et al.* 2006).Here we assess the impacts of the AES treatments on pollinator abundance, diversity and community structure at a range of scales using a variety of methods.

7.1 **Objectives**

We had four overall objectives: i) to assess the effectiveness of establishing a range of ELS options aimed at providing floral resources for pollinators; ii) to link the abundance of floral resources to the overall abundance of farmland pollinators using a range of monitoring techniques; iii) to link the abundance of specific pollinator taxa to processes mediated by local (e.g. floral resources) and landscape (e.g. habitat context) scales; iv) monitor effects of treatments on population growth of *Bombus terrestris*.

7.2 **Transects & visitation**

7.2.1 Methods

We measured bumblebee and butterfly abundance on 4 occasions (May to August) each year. Foraging bumblebees and butterflies were counted along a fixed 2m $\times$ 50m transects in the centre of each treatment patch (CC= field edge to crop ×2; ELS = field edge to crop, natural regeneration, Tall Bird; ELSX= Tall bird, Deluxe bird, Bumblebird, Pollen& Nectar,Wildflower), and the plant species on which they were foraging was noted. Butterflies on the wing are also recorded. In addition two 2m×2m quadrats at each end of

the transect were observed for 2 minutes on each occasion and all insects measuring ≥3mm and the flower 'units' they visited within the quadrat were recorded and identified to species or species groups. Observations were carried out between 10.00 and 17.30 during dry weather and when the ambient temperature was above 13°C with at least 60% clear sky, or 17°C under any sky conditions.

7.2.2 Results

7.2.2.1 Dicotyledon flower abundance.

On average the ELS and ELS Extra treatments resulted in significantly higher abundance of flowers (between three- and twelve-fold increases respectively when compared to CC (Repeated Measures ANOVA, $F_{2,8}$=22.56; P<0.001, Figure 7.1). Of the flowers actually visited by insects across the landscape there was a between three and ten-fold increase in the availability of floral resources compared with cross compliance (Repeated Measures ANOVA, $F_{2, 8}$= 27.92 P <.001) showing that nectar and pollen sources had been greatly increased in these landscapes. No year × treatment interactions were significant.

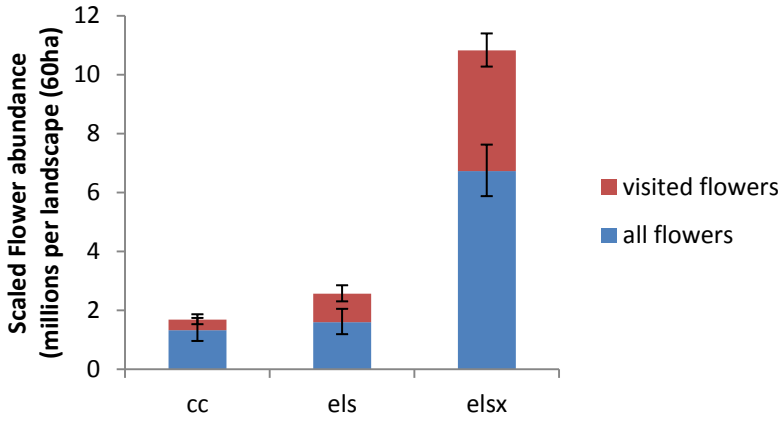


Figure 7.1. Mean flower abundance pooled across different options (see text for details) and scaled to represent floral resource availability at the scale that treatments were applied. Error bars are 1se.

7.2.2.2 Total pollinator abundance

Total pollinator abundance (bumblebees, solitary bees, honey bees and hoverflies) was 2 and 10 times greater in the ELS and ELS Extra treatments respectively compared with Cross Compliance (Figure 7.2). This increase in abundance corresponds well with the increase in floral resources observed (Figure 7.1). Butterfly abundance on transects was 1.5 and 4 times greater in ELSX and ELS compared to CC (Repeated Measures ANOVA, $F_{2,8}$= 79.96 P<0.001). Bumblebee abundance on transects was 4 times higher in ELS and over 30 times higher in

ELS X when compared to CC (Repeated Measures ANOVA, $F_{2,8}$=*79.61* *P*<0.001) Neither bumblebee nor butterfly species richness were significantly affected by treatment.

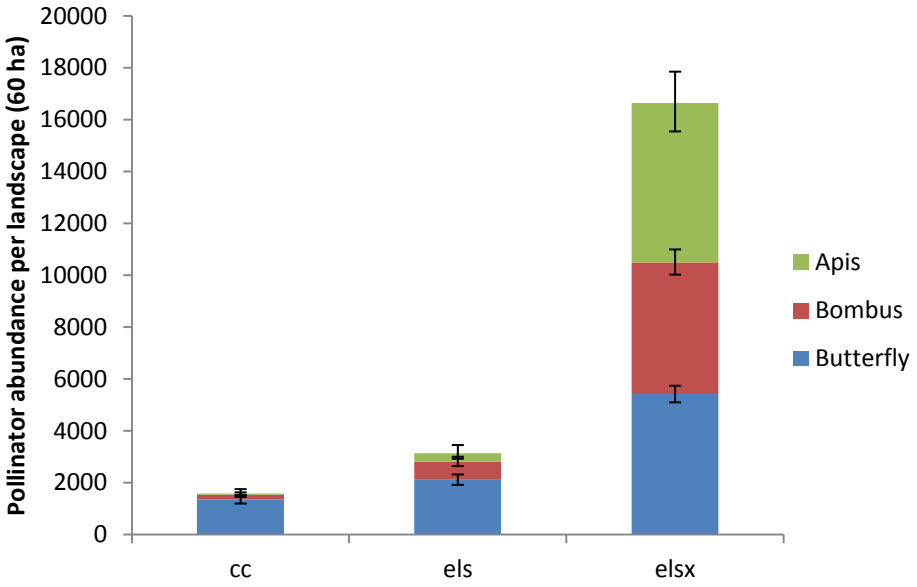


Figure 7.2. Mean abundance of butterflies, bumblebees and honeybees pooled across different options and scaled to represent abundance at the scale that treatments were applied. Error bars are 1se.

7.2.2.3 Flower-pollinator interactions

In terms of actual flower-pollinator interactions (measured using timed observations) there were between 7 and 60 times more in the ELS and ELS X treatments compared to CC in the years 2007 and 2008 (Figure 7.3). The majority of visits were made by syrphid flies (64%) and bumblebees (25%). Significant treatment effects were observed for *Apis* ($F_{2,8}$=22.49 , *P*<0.001), syrphids ($F_{2,8}$=32.1 , *P*<0.001) and bumblebees ($F_{2,8}$=17.37 , *P*<0.001). Solitary bees showed only significant year effects on visitation with much higher numbers in 2007.

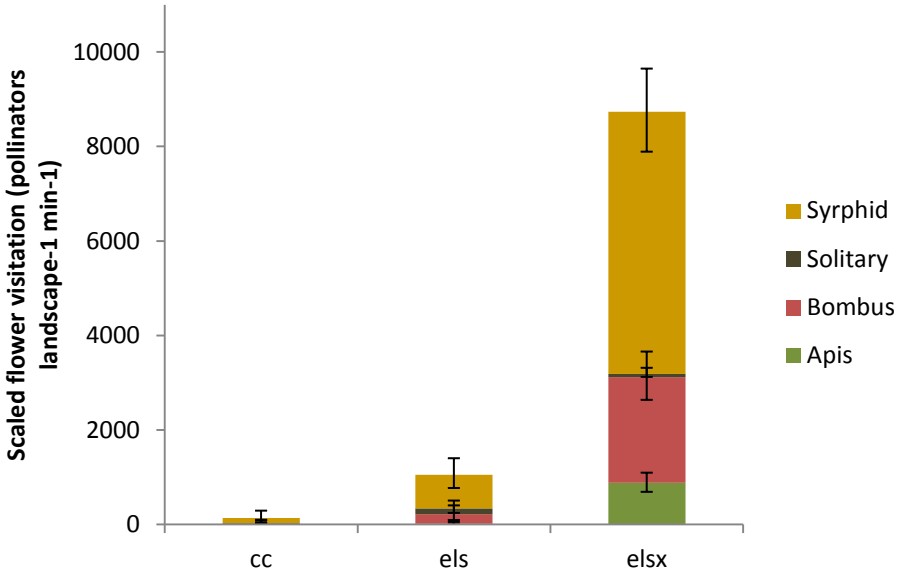


Figure 7.3. Mean (± se) number of visits (2007&2008)made by key groups of pollinators across different options and scaled to represent abundance at the scale that treatments were applied.

7.2.3 Discussion

In agreement with previous findings (Pywell *et al*. 2005) the abundance of bumblebees, hoverflies, solitary bees and butterflies were strongly related to flower abundance. These results suggest that high quality habitats providing a range of floral resources can act as a source for pollinating insects. However it is unclear from these data whether this was a behavioural effect, drawing in pollinators from the surrounding landscape, or a population dynamic effect, promoting or sustaining increased reproduction. The data to answer this question are few but it has been demonstrated that a highly targeted option like the pollen and nectar mixture (aimed at longer tongued bees) is able to promote population growth if targeted to landscapes that have few resources (Heard *et a.l* 2010). Our data show that the effect of flower rich options in the ELS X treatment had a disproportionate effect on visitation and abundance. It is clear that for the full effect of these options to be obtained high quality must be maintained. In the case of the pollen and nectar mixture this proved difficult with floral resources declining rapidly after year 2 due to the mortality of some of the key legume species. This effect occurred in the absence of competition from grasses and necessitated the rotation of patches across the experiment. In contrast the wildflower corners maintained and in some cases increased the provision of flowers. Although generally lower in floral density the greater flower species richness of this option provides a greater range of resources for oligo- and polylectic species.

7.3 **Trap nesting bees, wasps and their parasitoids**

The way in which different species, functional groups and trophic levels perceive different habitats and landscapes can lead to variation in how they interact (Tscharntke & Brandl 2004; Tylianikis 2005). It has been suggested that higher trophic levels, especially those with specialized niches like parasitoids, may be more sensitive to habitat context than their invertebrate hosts. This results from them having to locate habitat patches occupied by suitable hosts, in contrast to their hosts, who often have less specific requirements, and only have to find suitable habitat patches (Kruess & Tscharntke 1994). This variation in the sensitivity of parasitoids and hosts to landscape context may lead to situations where hosts are effectively 'released' from parasitisim (Kruess & Tscharntke 1994; Roland &Taylor 1997) which in turn may affect the stability of populations and the functions they perform.

In this study we examined how ELS enhanced habitats and the knock on effects on overall landscape composition and configuration affected nest colonization of solitary bees, wasps, and their parasitoids. Solitary bees provide important pollination services and solitary wasps can provide biological control services. Collectively they require a diverse range of floral and other food resources and nesting structures (both above and below ground) which makes them sensitive to plant species richness, habitat composition and landscape context. We focused on cavity-nesting bees (Hymenoptera: Apoidea) and wasps (Sphecidae, Eumenidae, Pompilidae) which are mainly solitary species that construct nests above ground in non crop habitats and make use of cavities and holes of varying diameter (3 – 8mm) in plant stems and crevices. The female constructs a series of cells (constructed from different materials according to species characteristics), each with one egg, which are provisioned with food for the larvae. This consists of pollen and nectar for bees and insects or spiders for wasps. Parasitoids or natural enemies belonging mainly to the Hymenoptera may lay their eggs in the cells, and their larvae consume the host and its' food.

We tested the following hypotheses: i) Nest colonization by bees and wasps increases with increasing proportion of non-crop habitats (landscape composition). ii) Parasitoids are more affected by agricultural intensification at local and landscape scales than their bee and wasp hosts.

7.3.1 Methods

We constructed standardised nesting sites (trap nests) to assess the effects of treatments and landscape on colonization and productivity of cavity-nesting bees, wasps and their parasitoids. Pairs of trap nests, each consisting of 20cm long internodes of reed (x300–500) and bamboo cane (x8) within a 13cm diameter plastic tube, were established on posts along field boundaries in the centre of each treatment unit (a total of 30 traps per annum: 3 treatments x 5 blocks x 2 traps) in spring. Trap nests were retrieved in autumn each year and opened. Each occupied reed or cane (henceforth "nest" was categorised according the type of material used by the female to divide cells and plug the nest entrance as follows: leaves (used by leaf-cutter bees), mud (used by mason bees, potter and spider-hunting wasps), cellophane (used by some bee genera) and 'other material' including various plant fibres. All occupied nests were individually bagged for rearing. After an initial chilling period, diapause was accelerated by gently warming in controlled temperature through the winter months. All emerging adults were identified to species.

General linear models were used to assess the response of overall abundance, overall Shannon Weiner diversity (H') and abundance and Shannon Weiner diversity (H') of bees, wasps and parasitioids to treatment (CC, ELS, ELS X) year and landscape structure. A Poisson distribution and a log-link were used for counts and models were re-fitted with a quasipoisson adjustment where over-dispersion of variance was high. Diversity was treated as normally distributed. The pollen and nectar margins and wildflower margins were considered as one combined landscape element as they provide similar floristic resources at a landscape scale. This approach also avoids proliferation of inter-correlated variables that explain landscape structure. Tussock grass margins and natural regeneration margins were similarly combined and treated separately as permanent grass cover as they are structurally distinct being dominated by tussock forming grasses (e.g. Dactylis glomerata). The proportion of each of these categories within the 500m radii surrounding each sampling plot was calculated. This radius was chosen according to the results of previous research on trap-nesting bees and wasps (Steffen Dewenter 2002, Holzschuh 2010). Each model tested the significance of block, treatment, year and the landscape parameters ($H_{Landscape}$), and the proportional cover of crop, permanent grass cover, hedgerows, tussock and natural regenerated grass field margins, floristically diverse field margins (wildflower and pollen and

nectar). In each case the most parsimonious model was identified by stepwise removal of non-significant interactions and main effects.

7.3.2 Results

A total of 2956 nests were found within a total of 118 traps over all four years (2 traps were damaged during 2006). In total 5019 bees (13 spp.), wasps (30 spp.) and parasitoids (26 spp.) were identified. We plotted the empirical cumulative distribution for all data across all years to compare species abundance distributions. This is similar to more traditionally used rank abundance diagrams but standardises the axes by total species richness and total abundance to overcome the problem of lack of independence where species richness differs. The slope is indicative of evenness and suggests that the bee community is more dominated by fewer species than the wasps and parasitoids. In this case *Hylaeus communis* constituted >70% of the abundance. The intercept of inflexion indicates the fraction of rare species and suggests too that there were relatively more rare bees.

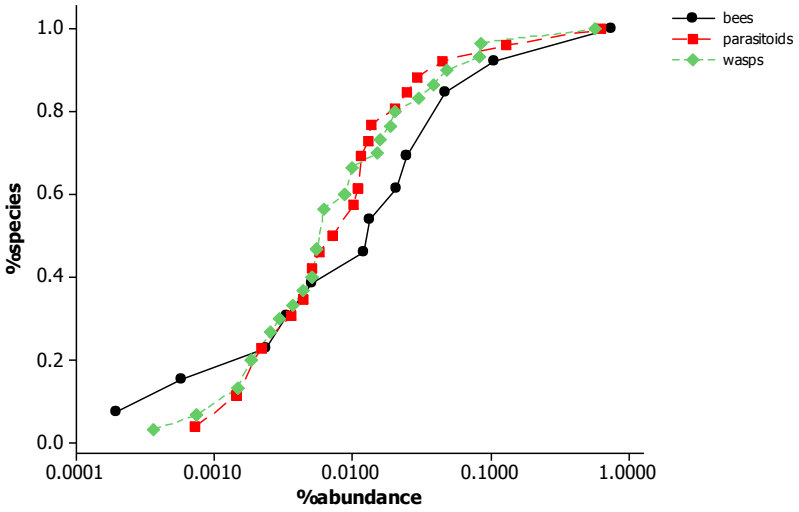


Figure 7.4 Comparisons of aculeate and parasitoid species abundance distributions across the Hillesden experiment (2006-2009) plotted on an empirical distribution function plot.

Overall differences in types of nest plug construction using showed significant differences across years for leaf and cellophane plug types (P<0.05) with a noticeable peak in cellophane plug types in 2007. A significant effect of treatment was shown only for nests with mud plugs ($F_{4,98}$ =3.94, P=0.006) with greater colonization in the ELS extra treatments than in Cross Compliance (Figure 7.5). This is suggestive of a positive effect of sown mixtures

and sward architecture on both pollen and nectar and invertebrate food items for bees and wasps respectively.

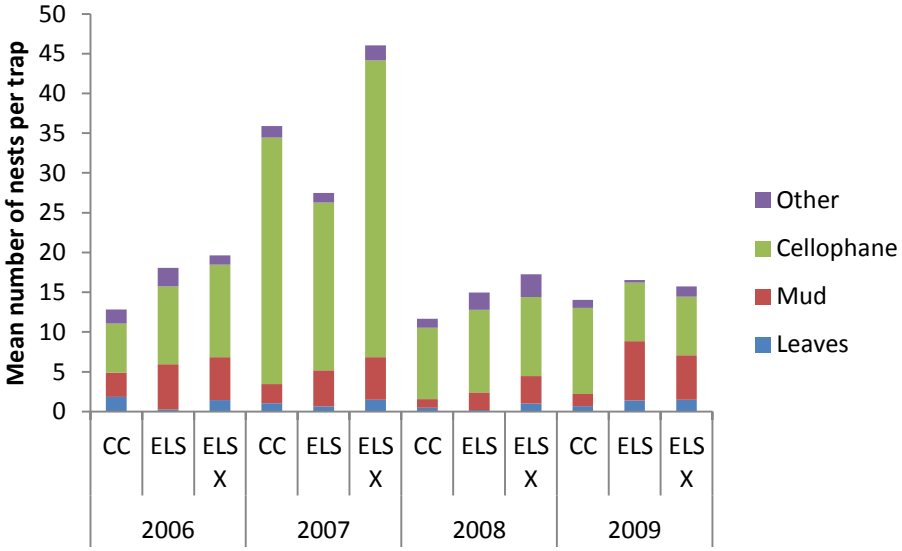


Figure 7.5. Number of colonized nests according to plug type

Overall species diversity, based on all the individuals that emerged from the trap nests, was significantly higher in the ELSX treatment (Figure 7.6, $F_{3,53}$ =3.23, *P*=0.048). There was no significant effect of year despite a tendency in some years for greater colonization in the ELS extra than ELS and Cross Compliance treatments. Focusing on diversity within individual groups revealed no significant effects of treatment, although wasps showed a significant effect of year ($F_{3,47}$ =6.29, P=0.001) with increases in the first three years and a decrease to the lowest value in 2009.

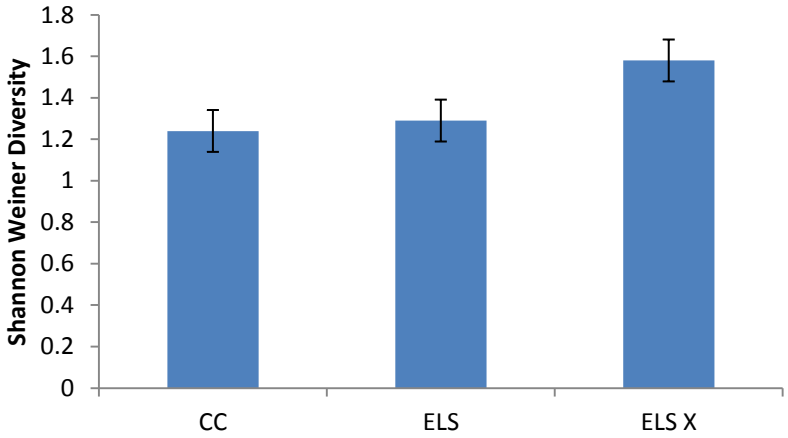


Figure 7.6. Mean Shannon Weiner Species diversity across all species and years per main treatment. Error bars are approximate ± 1 estimated standard error based on residual deviance.

Abundance was significantly increased ($F_{3,52}$ =4.5, P=0.007) over the first three years of the experiment but was lowest in the last year of monitoring (Figure 7.7).

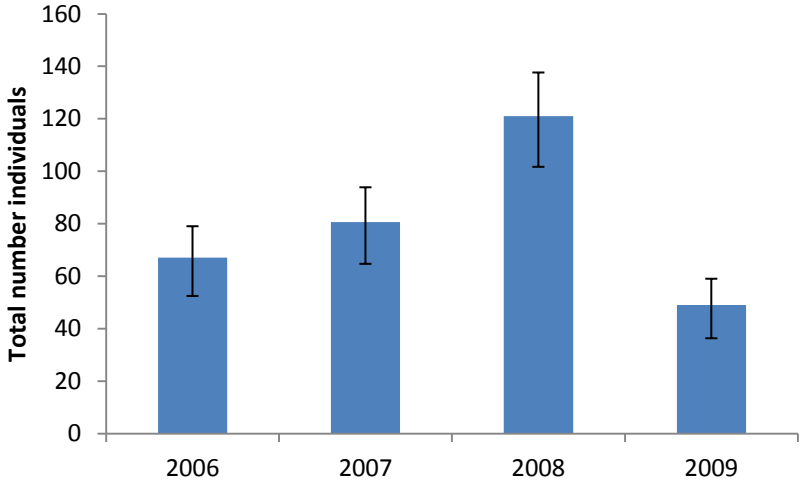


Figure 7.7. Mean Shannon Weiner Species diversity across years per main treatment. Error bars are approximate ± 1 estimated standard error based on residual deviance

The total abundance of bees showed significant effects of both year ($F_{3,50}$ =5.13, P=0.004) and treatment($F_{2,50}$ =4.55, P=0.001). The effects of year were similar to those for all individuals grouped whereas the effects of treatment showed significantly lower abundance in the ELS treatment (Figure 7.8)

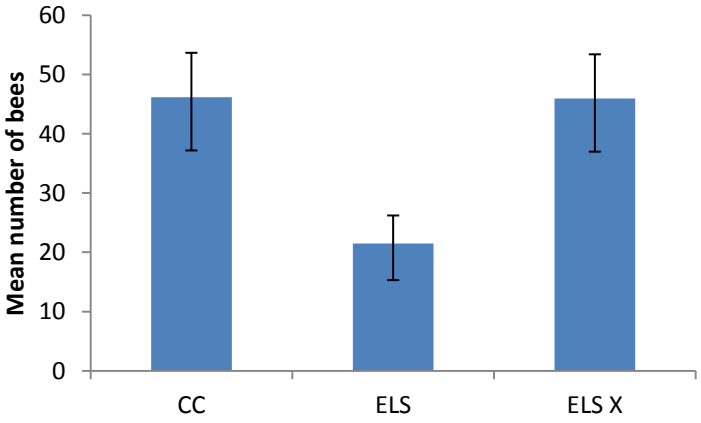


Figure 7.8.Effects of treatment on mean numbers of bees based on GLM with quassi- poisson error structure. Error bars are 1 se.

Wasp abundance was also significantly affected by treatment ($F_{2,50}$ =4.55, P=0.001) with significantly higher numbers emerging from traps in the ELS and ELS X treatment.

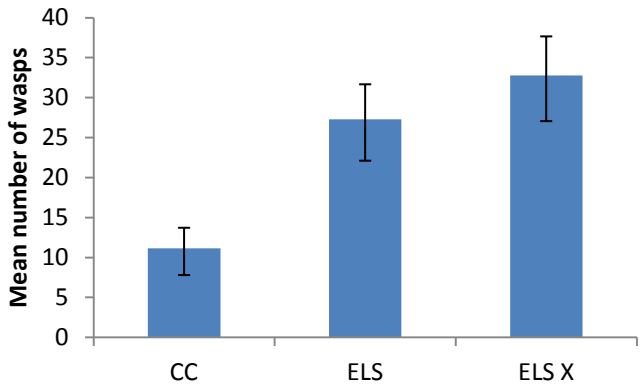


Figure 7.9. Effects of treatment on mean numbers of wasps based on GLM with quassi- poisson error structure. Error bars are 1 se.

Abundance of parasitoids was significantly affected by year and showed similar patterns to wasps ($F_{1,51}$ =4.55, P=0.035). However the effects of local landscape were mediated by effects on host abundance and distribution. Species richness and number of parasitoids increased with increasing host species richness ($F_{1,59}$ =40.4, P<0.001; species richness $_{(Parasitoid)}$ =0.434+0.452 × species richness $_{(Wasps+Bees)}$, r^2=54.4%) and increasing host abundance ($F_{1,58}$ =21.75, P<0.001; Figure 7.10).

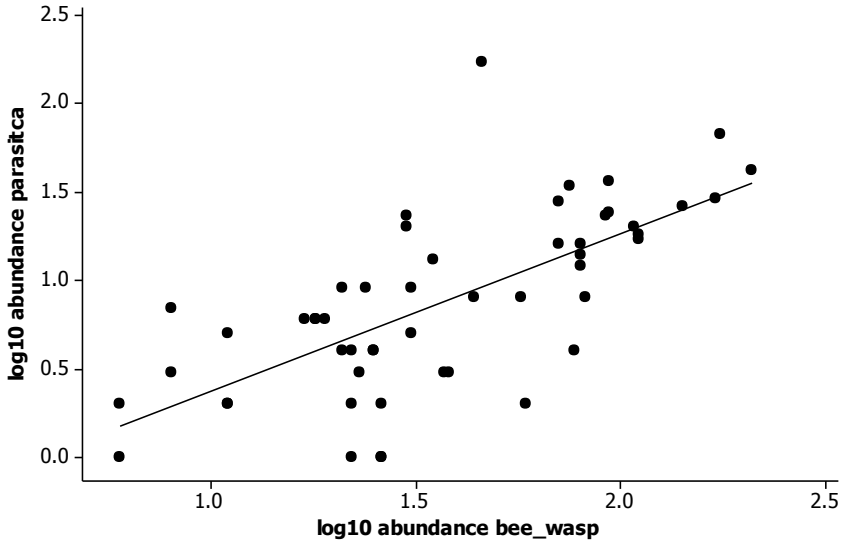


Figure 7.10. Effect of host abundance on parasitoid number. Regression equation , log10 abundance $_{parasitca}$ = - 0.523 + 0.894 log10 abundance $_{bee\_wasp}$ r^2=46.2%.

The significant effects of treatments disappeared when landscape variables (proportion of sown flower margins, proportion crop, proportion hedgerow, proportion grass margin within 500m radius) were included in the models (with block, treatment, year).

Overall diversity was significantly correlated with the proportion of sown flower rich margins ($F_{1,59}$ =7.3, P=0.009) and negatively correlated with the proportion of grass margins ($F_{1,59}$ =6.99, P=0.011). Overall abundance was positively correlated with proportion of crop ($F_{1,59}$ =7.29, P=0.011) and proportion of sown flowers ($F_{1,59}$ =7.19, P=0.009). Similarly diversity of bees was positively enhanced by flower rich margins ($F_{1,59}$ =7.3, P=0.009) and crop ($F_{1,51}$ =17.18, P<0.001) although the effect of treatment remained significant. Bee abundance was correlated with flower rich margins, crop proportion and proportion of flower rich margins. Wasp diversity decreased with increasing proportion, half a species was lost with each 10% increase in grass margin proportion ($F_{1,48}$ =11.15, P=0.002). In terms of abundance both proportion hedges and proportion crop had positive effects. Parasitoid species richness was weakly positively correlated with proportion crop ($F_{1,59}$ =5.12, P=0.027, r^2=0.08).

7.3.3 Discussion

The results of our study support our hypotheses that factors at all spatial scales, including AES treatment and habitat composition determine nest colonization patterns. While cavity-nesting bees and wasps depend on nesting sites in non-crop habitats, they utilise multiple habitats for foraging. Their foraging habits are not fully understood and the relative importance of landscape composition depended on the species group tested.

Overall diversity was significantly higher in ELSX because of high proportions of flower rich, non-crop habitat. However this composite effect on species diversity was not reflected in significant effects of treatment on diversity at the group level. Instead there were strong positive effects of sown flower cover on bee diversity and negative impacts of grass margin cover on wasp diversity. Strong year effects were observed and across the whole farm abundance and diversity of all groups increased for the first three years of the experiment. However it is clear that high mortality may have occurred over the winter in 2008/09 due to unusually low temperatures.

Bee abundance was enhanced by high proportions of flower rich, non-crop habitat, whereas wasps were enhanced by proportion of hedges and crop. The positive effect of hedge cover rather than of total area of non-crop habitat for nest colonization by wasps suggests that they may have provided connectivity and facilitated wasp movements between trap nests and source habitats. Wasps prey on a range of invertebrates e.g.

spiders, aphids and lepidopteran larvae, which are used for nest provision and are provided by different elements of the landscape including the crop. Our results show that even non-crop nesting specialists do not generally perceive crop fields as hostile. In contrast, bee abundance may have been enhanced by increasing the overall abundance of floral resources, and number of non-crop habitat in the landscapes, independent of connectivity. The differing response of bees and wasps to landscape factors may have consequences for pollination and predator–prey systems. While bees may provide pollination services within at least 500 m radius around their non-crop nesting habitats, predation by wasps may decline and predator–prey interactions may shift in favour of prey in landscapes where boundary structures are reduced.

There are two ways that food availability may influence the abundances in trap nests: i) enhanced reproduction rates through reductions in the costs of foraging for larvae where food availability is high; ii) enhanced colonisation if females prefer nests in the vicinity of floral resources or abundant prey (Klein, Steffan-Dewenter & Tscharntke 2004, 2006; Albrecht *et al*. 2007). Higher abundance of offspring can be explained either by a high number of females colonising the trap nest or by higher reproduction rates. In contrast, high species diversity can only be explained by a high number of females colonising trap nests, but not by higher per female reproduction rates. The increase in abundance in both bees and wasps suggests that females prefer to nest nearer abundant forage resources, presumably because it reduces the costs of provisioning nests.

Parasitoid species richness and abundance were well-explained by both host species richness and host abundance. This does suggest that landscape enhancements that promote greater heterogeneity are more effective at promoting more complete communities. Although the abundance of parasitoids was weakly positively affected by increasing proportion crop this is most likely to reflect the concomitant decrease in intensive grass cover which, when highly grazed as around Hillesden, provides little additional/alternative prey. It may also suggest that parasitoids are less sensitive to agricultural intensification and reduced habitat quality than their hosts. Parasitism rates in trap nests did not decrease in smaller and more isolated orchard meadows in central Europe (Steffan-Dewenter & Schiele 2008) and even increased with increasing agricultural intensification in Ecuador (Tylianakis

2007). Our results underline that changes in land use intensity and habitat quality can result in changed trophic interactions, but the direction of changes are not generally predictable.

7.4 **Pan traps**

Pan trapping is a good passive sampling method for assessing insect pollinator communities since it collects high species numbers, is largely unaffected by collector bias and is quantitatively similar to transect methods (Westphal *et al.* 2008). We used two approaches to this study: i) white traps (most suitable for aculeates, Bees Wasps Ants Recording Scheme Handbook) placed centrally to each treatment each June, July and August to measure general response of key groups to treatments and landscape; ii) a focused study on all pollinators using 3 UV colours to consider the widest spectra of pollinators with the specific aim of understanding body size patterns.

7.4.1 Pan traps – temporal trends

7.4.2 Methods

Here we considered how AES treatment and the structure of the landscape influenced the abundance of pollinator groups in an arable landscape across time. For each sampling round six white pan traps were placed 5 m apart along a margin at the central point of each treatment (15 locations). Individual pan traps were 20 cm in diameter and had a depth of 5 cm. Each pan trap was half filled with water and a few drops of unscented dishwashing detergent were added to reduce surface tension. Traps were left out for 48 hours in June, July and August of each year (2007-2009). Note trapping was done in 2006 but only in June so these results have not been included here. The contents of pan traps were sorted and identified to broad groups defined as: syrphids, other Diptera, *Bombus* spp, solitary wasps, other Apidae spp., other hymenoptera (including parasitica), Lepidoptera and total invertebrates. For the purpose of this analysis pan trap samples were summed across monthly sampling dates and the six traps within a particular experimental unit. Data were analysed using generalised linear models with a quasi-Poisson error distribution (to account for over-dispersion in the data) and a log link function. Block, treatment and year were included as factors along with continuous landscape variables: proportional cover of all crops; proportional cover of sown flower margins; proportional cover of grass margins; proportional cover of hedgerows; Shannon Weiner habitat diversity. Minimum adequate

models were constructed by stepwise addition of the most significant factors until the addition of further explanatory variables resulted in a non-significant increase in the explanatory power of the model. The significance of two way interactions between these explanatory variables were tested using the same approach.

7.4.3 Results

Over 286,886 invertebrates were counted from the pan traps. This was dominated by non pollinator species groups (64%) followed by other Diptera (24%), other hymenoptera (9%) and the other groups the remainder Syrphidae > Lepidoptera > Apidae others > *Bombus* spp.>solitary wasps.

In all groups there were significant effects of year on abundance (P>0.05). For *Bombus* this time effect ($F_{2,31}$=5.39, P=0.01) showed an overall increase in abundance and this was also true for other hymenoptera ($F_{2,36}$=15.08, P<0.01), solitary wasps ($F_{2,36}$=9.02, P<0.01) syrphids ($F_{2,36}$=9.02, P<0.01), total diptera ($F_{2,36}$=3.83, P=0.03), lepidoptera ($F_{2,34}$=13.63, P<0.001) and all invertebrates ($F_{2,36}$=5.43, P=0.009). Only other Apidae showed a decline ($F_{2,30}$=6.57, P<0.001) with catches halving over time.

Across all groups only four showed significant effects of treatment (Figure 7.11). *Bombus* showed a positive effect of ELS and ELS X when compared to CC ($F_{2,31}$=3.74, P=0.035). This was explained by a positive significant effect of proportion of sown flower margins ($F_{1,36}$=8.78, P=0.005) at the landscape scale. In the remaining groups higher densities were associated with the CC treatment. Other hymenoptera ($F_{2,31}$=5.62, P=0.007) showed densities similar in CC and ELS that were significantly higher than those in ELS X. This correlated negatively with an increase in sown flower margins and was significantly and positively with an increase in grass margins ($F_{1,37}$=8.11, P=0.007). Similarly syrphids ($F_{2,36}$=13.3, P<0.001) showed decreases in abundance with increasing AES intervention. However, at the landscape scale increases in abundance were associated with higher proportions of sown flowers in the landscape ($F_{1,35}$=5.53, P=0.024). Solitary wasps densities were higher in CC than ELS and ELS X ($F_{2,31}$=3.95, P=0.03) but at the landscape scale decreased with proportion of cropped land ($F_{1,31}$=6.87, P=0.14) and increased with proportion of sown flower margins ($F_{2,31}$=9.3, P=0.005).

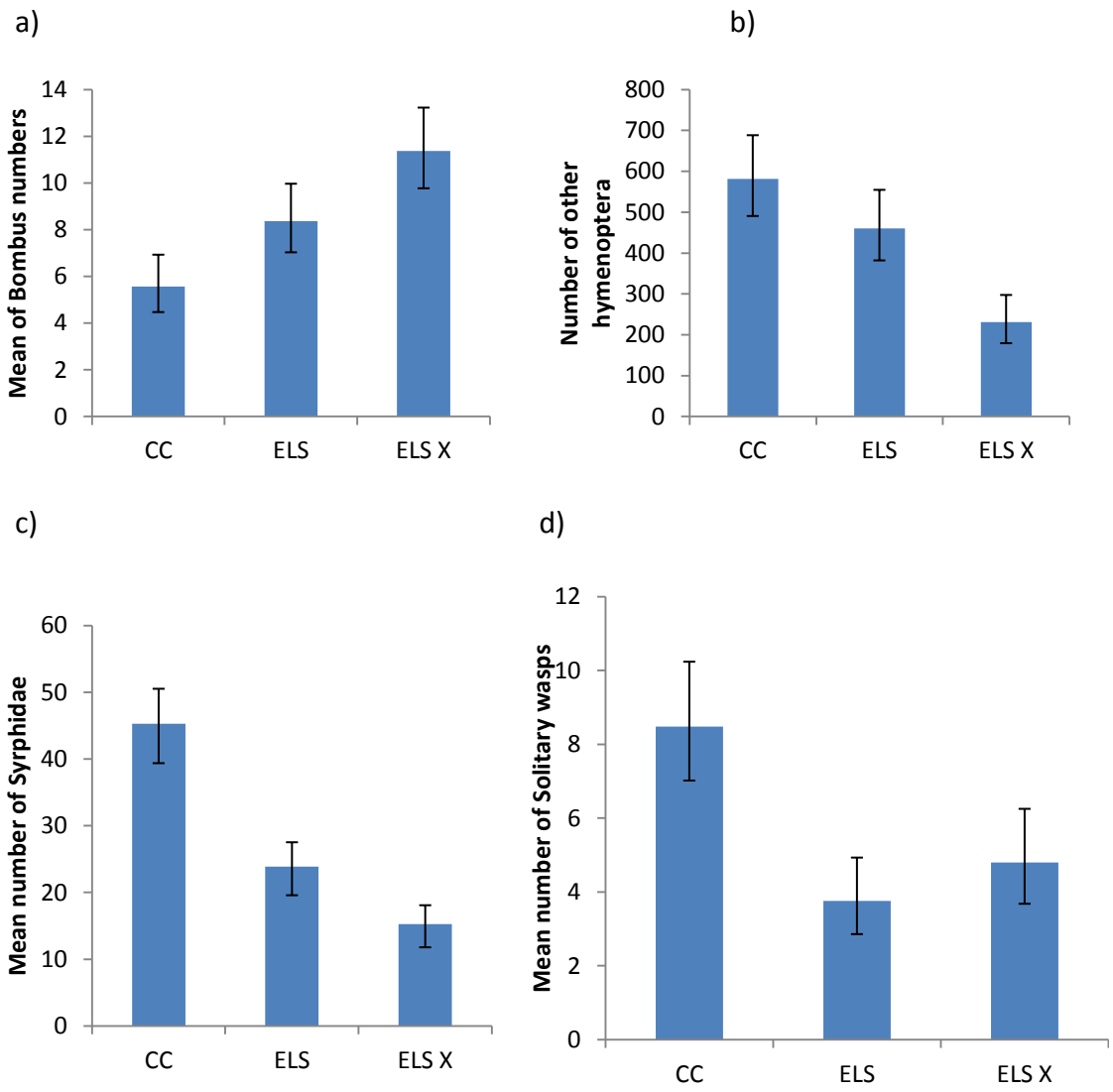


Figure 7.11 Mean numbers of a) *Bombus* ,b) other hymenoptera, c) syrphidae and d) solitary wasps caught in pan traps. Error bars are 1 s.e.

7.4.4 Discussion

The significant positive effects over time suggest that at a whole farm level abundance of a range of pollinator types showed a general increase. However the decrease in the last year of monitoring in may groups (in common with effects seen in stem nesting aculeates) demonstrates that large scale shifts in abundance are possible (perhaps due to adverse weather conditions like the several harsh winters seen through the project and subsequent higher mortality rates). An interesting question would be to understand if ELS is able to improve the resilience of pollinator communities.

Overall there were significant effects of treatment but some of these appear counter to results from transect counts and observations. For example hoverflies were observed to visit more flowers in ELS X margins but with pan traps the counter effect was apparent. At

the wider landscape scale the positive effects of flower provision in margins became apparent. In the case of hoverflies, Branquart & Hemptinne (2000) showed that many species show only a limited preference for flower species, instead targeting flowers with large inflorescences and flat corollae (e.g. Asteraceae). Similarly, butterfly habitat generalists are far more common in intensively managed agricultural landscapes (Ekroos *et al.* 2010). Such generalist species would be more likely to be represented in most margin or crop types than more specialist species. It is also likely that simple measures of community structure, like abundance, collected passively with pan traps do not necessarily reveal underlying changes in the composition of species associated with each management treatment and their interactions. In addition the effect of bias in catches cannot be totally discounted. The attractiveness of pan traps (which mimic floral resource abundance) is relative to the local context. Pan traps in the vicinity of flower rich reserves may be perceived as lower quality than where floral resources are scarce.

7.4.5 Pan Traps - body size distribution

Here we considered how local habitat type and the structure of the landscape influenced the individual body size distributions of pollinators in an arable landscape. While direct measures of foraging ranges are limited to only few pollinator species (e.g. Osborne *et al.* 2008), body size represents a functional trait that can be used to predict dispersal distances (Haskell *et al.* 2002; Greenleaf *et al.* 2007). The distribution of body sizes may therefore provide key insights into the capacity of pollinator communities to both forage and persist within complex agricultural landscapes (Bommarco *et al.* 2010). While larger bodied species may be better able to both move between isolated resources and so colonise new habitat patches (Hanski & Ovaskainen 2000), they may conversely be more prone to increased predation as well as show a greater likelihood of stochastic extinction events (Ewers & Didham 2006). In the context of this study individual size distributions (also called size spectra) were used to describe the distribution of pollinator biomass within sequential body size classes (irrespective of species) (Blumenshine *et al.* 2000).

7.4.6 Objectives

We tested the hypothesis that larger bodied pollinators are better able to utilise spatially isolated resources and thus more likely to colonise isolated habitat patches (Hanski & Ovaskainen 2000). Therefore body size distributions will shift towards larger bodied

pollinators where there are low proportions of semi-natural habitats. In addition, we expected the body size distributions of pollinators to be modulated by other functional traits linked to the utilisation of resources by pollinators within the wider landscape, i.e. larval development and sociality (Bommarco *et al.* 2010; Öckinger *et al.* 2010).

7.4.7 Methods

For each sampling round six pan traps, two each of colours white, yellow and blue, were placed within each of the 35 different habitats. Two of these habitat types were conventional crops, winter wheat (*Triticum aestivum* L.) and oilseed rape (*Brassica napus* L.). The remaining five habitats were margins: 1) natural regeneration; 2) tussock grass; 3) pollen & nectar; 4) wild bird margins; 5) wildflower margins. The pan trap colours were chosen to mimic the prevailing colour of flowers in the sample region(Westphal *et al.* 2008). In each case UV-bright white, yellow and blue paint was used to improve the attractiveness of the pan traps to pollinators (Sparvar Leuchtfarbe, Spray-Color GmbH, Merzenich, Germany). Three pan traps (one of each colour) were arranged in a triangle 1.5m from each other, with a second group of three traps positioned 5m from these. Individual pan traps were the same size as above. Traps were run for a period of 72 hours on two sampling dates, starting on 10/6/2008 and 14/7/2008. The contents of pan traps were sorted and identified to species. Pollinators were defined as those species that play a key role in the transfer of pollen between flowering plants (Forup & Memmott 2005). These included bees (Hymenoptera: Andrenidae, Apidae, Megachilidae, Halticidae), butterflies (Lepidoptera) and hoverflies (Diptera: Syrphidae).

Body size was used as a surrogate for functional traits linked with foraging range / dispersal distances (Greenleaf *et al.* 2007). The mean body lengths of each species were determined by direct measurement of ten individuals, with dry mass determined using Rogers *et al.* (1976) length vs. body mass correlation. This length *vs* mass correlation represents a cross taxa relationship for insects, including the orders Lepidoptera, Hymenoptera and Diptera. This correlation has an R^2 of 0.94 and is thus robust. The ratio of mass to wing area scales linearly with body dimensions for most insects, and as such body mass provides a useful basis for making inferences about dispersal (Dudley 2002).

Normalised size spectra were used to characterize the distribution of pollinator biomass within each of the experimental plots (Blumenshine *et al.* 2000). These were

created by plotting the value $\log_2$ (sum of biomass in each size class/range of the size class) against the maximum body size in each body size class on a $\log_2$ scale. By comparing the slopes of these normalised size spectra we were able to investigate how the structure of body size distributions changed in response to local management and landscape structure (Blumenshine *et al.* 2000). A slope equal to -1 indicates that the distribution of total biomass across each of the $\log_2$ size classes is equal (i.e. each size class, representing a doubling in size, contains the same biomass of pollinators). Where slopes were less than -1 the distribution of total biomass was skewed towards the smaller body size classes. Conversely where the slopes were greater than -1 the distribution of biomass was skewed towards larger body size classes. By looking for systematic responses in the slopes of normalised size spectra, changes in the body size distributions of pollinator communities in response to underlying environmental gradients can be investigated. The slopes and standard errors of normalised size spectra were calculated for each of 35 habitats. This was done for all pollinators together, as well as separately according to whether pollinators were social or non-social (sociality) and whether the larvae were typically above ground (aerial) or on the soil surface / below ground (subterranean). Larval location refers to nests in the case of bees, or free living larvae in the case of hoverflies and butterflies. These traits have been shown to be important in predicting pollinator responses to landscape structure (e.g. Steffan-Dewenter & Tscharntke 2000; Bommarco *et al.* 2010; Williams *et al.* 2010).

Pan trap samples were summed across both sampling dates and the six traps within a particular experimental unit. General linear mixed models were used to test the response of species richness ($\log_e$ N+1 transformed) and normalised size spectra slopes for all pollinators were against the explanatory variables of: 1) local management type (e.g. oilseed rape, tussock margin); 2) proportional cover of all crops; 3) proportional cover of mass flowering crops; 3) proportional cover of sown field margins; 4) proportional cover of semi-natural grassland; 5) proportional cover of hedgerows; 6) habitat heterogeneity. Block was included as a random effect in all models. Minimum adequate models were constructed by stepwise addition of the most significant factors until the addition of further explanatory variables resulted in a non-significant increase in the explanatory power of the model. Following this, the significance of two way interactions between these explanatory variables were tested using the same approach. Degrees of freedom were calculated using the

iterative Kenward-Rogers method (Schabenberger & Pierce 2002). In order to account for variation in the explanatory power of each normalised size spectra slope, we weighted the general linear mixed models by the inverse of the standard error of these regression slopes (Blumenshine *et al.* 2000). The inversed standard errors themselves were not correlated with any of the explanatory variables ($p>0.05$), indicating that there was no systematic bias in the weighting of these mixed models.

Normalised size spectra were also derived separately in each experimental plot according to pollinator sociality (social or non-social) and larval location (aerial or subterranean). The effects of sociality and larval location on the response of normalised size spectra were analysed in separate mixed models. The same core set of fixed effect explanatory variables (as described above) were tested using stepwise addition, along with a fixed effect describing the functional traits (larval location or sociality) and the pair wise interaction of the trait with the other fixed effects. A significant interaction between a trait and a measure of landscape structure indicates that the distribution of body sizes is modulated by whether or not pollinator species share that trait. Block was included as a random effect, as was a treatment × block interaction. This treatment × block random effect accounted for the split-plot phenomenon that resulted from the fact that separate normalised size spectra slopes for each level of a trait were derived from pollinators collected within the same experimental plot.

Although honey bees (*Apis mellifera* L.) are effectively domesticated their response to landscape structure may differ from that seen by wild pollinators (Williams *et al.* 2010). However since they were relatively uncommon (6 % of total abundance) the decision was made to retain them in all presented analyses.

7.4.8 Results

A total of 77 species of insect pollinators, representing 1,106 individuals, were collected from the pan traps in 2008. While the hoverflies were the most numerically abundant of the pollinators, they represented a much smaller fraction of the total biomass. There was a *c.* 100 fold increase in body mass from the smallest (1.22 mg) to the largest pollinator (126.0 mg).

Pollinator species richness was positively correlated with habitat heterogeneity ($F_{1,33}$=5.94, p=0.02; Figure 7.12), while no significant effects of local habitat type or any of the other measures of landscape structure were found.

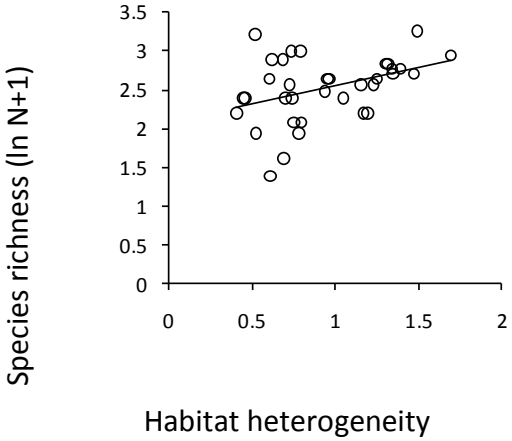


Figure 7.12 Response of pollinator species richness to habitat heterogeneity within 0.5km radii surrounding each sampling point.

The slopes of the normalised size spectra for all pollinator species considered together were positively correlated with both the proportion of semi-natural grassland ($F_{1,27.5}$=4.74, p=0.04; Figure 7.13a and habitat heterogeneity ($F_{1,8.8}$=6.56, p=0.03;Figure 7.13). Random effect covariance parameters were successfully estimated, although no other significant effects were found for either local habitat type or any other measures of landscape structure. The increase in the regression parameter of the normalised size spectra indicate that a greater proportion of the biomass of pollinators was found within larger pollinator size classes as the proportion of semi-natural grassland and habitat heterogeneity increased. However, in all cases the slopes of the normalised size spectra were greater than -1 indicating that the biomass of pollinators was always weighted (although to different extents) towards the largest body size classes. In no case were significant interaction terms found.

Normalised size spectra were also analysed separately according to the functional traits of larval location (aerial or subterranean) and sociality (social or non-social). Normalised size spectra showed no response to larval location ($F_{1,50.9}$=1.72, p=0.19) or any interaction of this factor with treatment ($F_{6,33.5}$=1.29, p=0.28) or of the measures of landscape structure (p>0.05). No other individual effects of treatment or landscape structure were found (p>0.05). This same pattern was repeated for the sociality trait. There was no significant effect of sociality ($F_{6,36.2}$=1.63, p=0.21), nor any interaction of this factor

with treatment ($F_{6,17.1}$=1.63, p=0.19) or the measures of landscape structure (p>0.05). Again, no other individual effects of treatment or landscape structure were found (p>0.05).

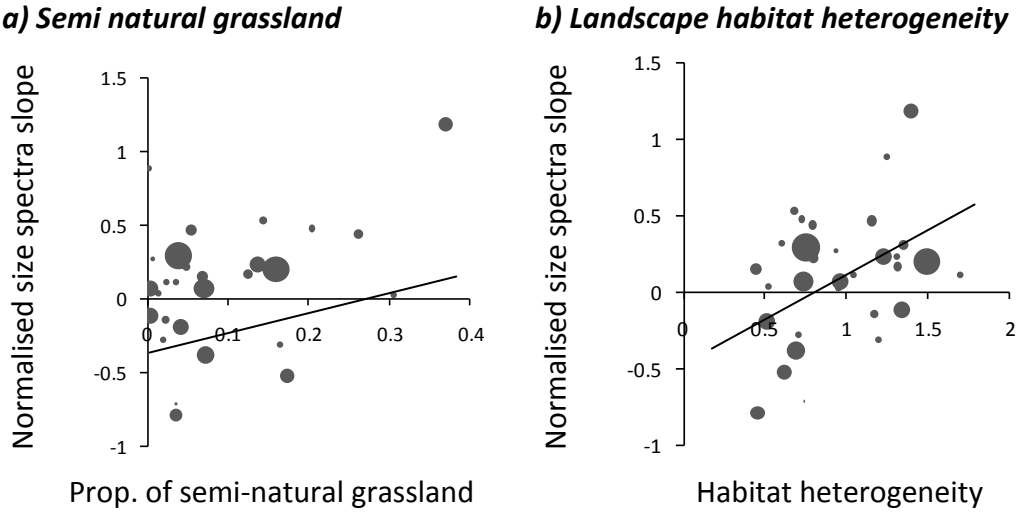


Figure 7.13. Response of the slope of the normalised size spectra for the pollinators in response to both the proportion of semi-natural grassland (A) and habitat heterogeneity (B) found within a 0.5km radii of each sampling point. Increases in the slopes of the normalised size spectra indicates an increase in the proportion of the total biomass of pollinators found within species of *larger body size classes. As the mixed models from which these correlation are derived were weighted by the inversed standard error of the normalised size spectra slopes*, the size of these slopes has been indicated by the area of each data point.

7.4.9 Discussion

Landscape structure is frequently reported as a key factor organizing pollinator communities (Weibull *et al.* 2000; Steffan-Dewenter & Kuhn 2003; Öckinger & Smith 2007; Jauker *et al.* 2009; Winfree *et al.* 2009; Ekroos *et al.* 2010). This dependence on landscape structure in part reflects the capacity of some pollinators to forage between widely spaced habitat elements (Greenleaf *et al.* 2007; Osborne *et al.* 2008; Jauker *et al.* 2009; Stevens *et al.* 2010). The heterogeneity of habitat types within the landscape was positively correlated with overall pollinator species richness. As many of the pollinator species have some degree of habitat specialization (Osborne *et al.* 1991; Steffan-Dewenter & Tscharntke 2000; Jauker *et al.* 2009) an increased density of habitat types would promote niche diversity and increase the chances of encountering more pollinator species at fixed sampling points (Weibull *et al.* 2000). Areas with high habitat heterogeneity were typically characterized by high proportional coverage of managed ELS margins, in particular flower rich field margins which are important for resource provision for a range of species (Heard *et al.* 2007; Ekroos *et al.* 2010).

In contradiction to our main hypothesis, we found that there was a shift in the distribution of pollinators away from large bodied species as the availability of semi-natural habitats in the landscape decreased. This was counter to our *a priori* expectation that large bodied species would be better able to utilise fragmented landscapes (Hanski & Ovaskainen 2000; Ewers & Didham 2006; Bommarco *et al.* 2010; Öckinger *et al.* 2010). One explanation for this effect is that resources tend to be have a fractal distribution, and smaller species are often more able to exploit finer-grained resources than larger species (Holling 1992). For this reason the areas over which species forage should scale more steeply with body size than would be expected from their resource requirements alone (Haskell *et al.* 2002). This effect is scale dependent, and as a result larger species may be less able to utilise the same density of resources as small species. Thus larger species are more prone to the effects of loss of key foraging habitats at a landscape scale (Haskell *et al.* 2002), potentially making them more sensitive to reductions in the area of semi-natural grassland (Harestad & Bunnell 1979). A similar pattern was also seen in pollinator body size distributions in response to increased habitat heterogeneity at a landscape scale. In this case we hypothesise that low habitat heterogeneity was associated with a reduction in the availability of flower rich habitat types, which impacted on the capacity of larger pollinators to obtain resources.

While the body size distribution responded to landscape structure, the nature of this response was not modulated by the traits of sociality and larval development location. Sociality was a trait found only among the bees in this study. Social bees have been suggested to be better adapted to utilising sparse patches of resources within intensively managed agricultural landscapes (Steffan-Dewenter & Tscharntke 2002). However, Bommarco *et al* (2010) failed to find an effect of sociality as a factor modifying bee species area relationships. Most of the social species found in this study (e.g. *Bombus* spp. and *A. mellifera*) were relatively large compared to other pollinators, while intra-specific size variation was a common factor in many colonies (e.g. *Bombus* spp.) (Bommarco *et al.* 2010). The failure to find an effect of sociality on the distribution of body sizes may in part be a result of these factors.

Where intensive agricultural management practices have been used, the abundance of bees nesting above ground has been shown to be lower than below ground nesting species (Williams *et al.* 2010). This is the result of above ground nesting species having a

greater exposure to damage caused by farm machinery. Similarly, butterfly larvae, which typically feed above ground, are also known to be sensitive to mortality resulting from agricultural management practices (Humbert *et al.* 2010). In the current study we included no measure of management intensity, nor would it have made sense to do so as the whole site was managed as a single unit. Larval location in the context of this landscape seems to have been of limited importance in predicting the distribution of pollinator body sizes. It is likely therefore that the same constraints on dispersal and foraging distances linked with body size apply to pollinators with aerial and subterranean larvae.

7.5 **Bumble bees colonies**

Resource availability and the presence of natural enemies are major components of habitat quality and can affect the fitness of many organisms. For central place foragers, such as social insects, the quality and distribution of food resources in the environment are particularly important, as foragers are limited to a restricted area around the nest (Dukas & Edelstein-Keshet, 1998). These include bumblebees, *Bombus* spp. (Hymenoptera: Apidae), primitively eusocial bees with an annual cycle that nest mainly in temperate habitats (Alford, 1975). Colonies require regular food supplies of nectar and pollen within foraging distance, and shortfalls in these, as well as the presence of parasites, can affect colony growth and reproduction (Bowers, 1985; Sutcliffe & Plowright, 1988; Cartar & Dill, 1991; Schmid-Hempel & Schmid-Hempel, 1998; Goulson *et al.*, 2002). To date most studies have investigated the effects of these factors in isolation, either under laboratory conditions or using artificial food supplementation in field colonies (Pelletier & McNeil, 2003). Here we investigated the effects of natural food availability on interactions between field colonies of *Bombus terrestris* (L.) and their social parasite, *B. (Psithyrus) vestalis* (Geoffroy).

Cuckoo bumblebees (sub-genus *Psithyrus*) are obligate social parasites of *Bombus* species. They tend to emerge from hibernation later than their hosts, have no worker caste, lack pollen collecting structures, and rely on host workers to rear their offspring. When a *Psithyrus* female invades a colony she often dominates the resident *Bombus* workers and may kill the founding queen, leading to reduced or failed reproduction of the host colony (Fisher, 1987). Nevertheless, the process of invasion and interactions between *Psithyrus* and *Bombus* can be highly variable (Fisher, 1988; Dronnet *et al*., 2005). *B. (Psithyrus) vestalis*

exclusively parasitizes colonies of *B. terrestris* and can effectively inhibit ovarian development in workers (Vergara *et al.*, 2003). In addition, founding queens can be usurped by late-emerging queen bumblebees of the same or different species attempting to take over established nests for themselves (Alford, 1975). Usurpation results in the death of at least one queen and, if successful, the invader continues to care for the original brood.

The success of both types of social parasitism may depend on the size and energetic status of a colony. Invasion attempts are thought to be more successful in small host colonies due to their low worker numbers and less effective defence than larger colonies (Fisher, 1984). Usurpation by *Bombus* queens, for example, rarely occurs following the emergence of second brood workers (Alford, 1975). Preferential invasion of larger colonies by *Psithyrus* has been observed in one study (Müller & Schmid-Hempel, 1992), although success was not measured in this case due to removal of *Psithyrus* soon after invasion. Food shortfalls or sparse foraging resources can also increase the vulnerability of colonies to predators and parasites. Energy-poor *Bombus* colonies exhibited less successful defence behaviour when attacked by intruding *Psithyrus* in a controlled laboratory experiment (Cartar & Dill, 1991) and food scarcity led to higher parasitism rates in the solitary bee *Osmia pumila* within flight cages (Goodell, 2003). Furthermore, bumblebee queens show considerable variation in their time of emergence from hibernation and subsequent colony founding. Founding a colony is itself likely to be stressful as queens must allocate resources to a variety of expensive activities, including the initiation of ovary activity, wax production and brooding behaviour (Alford, 1975). The timing of this phase in the colony cycle may further affect their susceptibility to parasitism. Early founding might, for example, lead to higher colony fitness in the absence of *Psithyrus*, but higher invasion rates where they are common (Müller & Schmid-Hempel, 1992).

The potential effects of local resource availability on social parasite activity, and on probabilities of nest invasion and success during the bumblebee colony cycle, have not been tested in the field. It is important to understand these aspects of the host-parasite relationship, especially in bumblebees, as they are now the focus of much conservation effort (e.g. Goulson *et al.*, 2005) and the diversification of farmland through agri-environmental policies could lead to changes in the availability of flowering resources in agricultural landscapes (Carvell *et al.*, 2007). The aims of this study were firstly to investigate

the effects of resource availability (AES treatment and cropping) on performance and social parasite invasion in field colonies of *Bombus terrestris*, and secondly to assess the influence of colony founding date on these parameters.

7.5.1 Objectives

We tested the hypothesis that colonies placed amongst abundant resources (ELS X or flowering rape fields) would grow larger and thus be better defended from intruders than those amongst sparse foraging resources (CC and wheat fields). We also tested if nests founded earlier in the season would be more prone to invasion by the social parasite *Psithyrus vestalis* or usurping *B. terrestris* queens.

7.5.2 Methods

We used laboratory-reared colonies of *Bombus terrestris* subsp. *terrestris* (Biobest Biological Systems, Belgium). We simulated early and late nest founding by placing two batches of colonies into the centre of each treatment unit: the first on 03/05/2006 and the second on 02/06/2006 (hereafter B1 and B2 respectively). Each batch consisted of 24 colonies with a founding queen and first brood of five to ten emerged workers (mean =7.7±0.48). Since the time taken for a queen to found a nest and raise a small number of workers is *c.* 4-5 weeks, the dates of colony placement corresponded with approximate initial colony founding dates of the end of March and end of April; well within the total period in which nest searching queens are typically observed (*unpublished data*).

Two bumblebee colonies from B1 were randomly assigned to the centre of each experimental treatment unit and positioned together (details on housing given below) along a sheltered field edge. We used the overall experimental design and the blocked nature of cropping (oilseed rape *vs.* winter wheat) to give 12 colonies surrounded by oilseed rape and 12 surrounded by wheat. Colonies from B2 were similarly placed in the field on 02/06/2006 at a distance of 20 metres from B1 colonies. This was considered to be relatively representative of the natural nesting situation in which colony densities are thought to be fairly high (Darvill *et al.*, 2004). The scale of our "field units" also meant that each set of colonies was separated by a minimum of 800 metres, exceeding the likely average foraging distance of *B. terrestris* workers (Darvill *et al.*, 2004). Thus, while workers from the colonies surrounded by wheat probably had to travel further to find forage (Westphal *et al*., 2006), it is unlikely that many would have reached the block of rape in our experiment.

Colonies were housed in ventilated plastic boxes within cardboard boxes. Each pair was protected from the effects of rain and temperature fluctuation by a larger polystyrene box and from damage by mammalian predators by a metal cage with 50mm mesh size. Prior to placement in the field, each colony was fed *ad libitum* with an artificial nectar solution and mixed pollen. This was continued for the first week in situ, as well as allowing workers to forage freely, to allow for adjustment to the new surroundings. The 16mm colony entrance hole allowed queens, workers and males to pass freely. Any *Psithyrus* and queens of other *Bombus* species that entered the boxes were left.

Colonies were assessed before being placed in the field and at weekly intervals thereafter. On each occasion, between 10:00h and 17:00h, we estimated the number of workers, recorded the condition of the foundress queen (which had been paint-marked on the thorax in the laboratory) and the presence and abundance of any invading *Psithyrus* or usurping *Bombus* queens. Observations continued until either the first sign of emergence of males (to minimise the escape of sexuals at the study site) or, if no reproductives were seen, until the entire colony had died out. At this time nest entrances were converted to prevent escape of bees from within the colony and capture all returning foragers. After 24h, colonies were removed from the field and immediately placed in a -80°C freezer. Subsequently we recorded the number of workers, males, queens (foundress and usurpers) and *Psithyrus* present in each colony.

7.5.3 Analyses

We tested for differences between colonies within each treatment and cropping regime (effects of resource availability), differences between batches (effects of timing) and interactions between resource type and batch. A repeated measures analysis of variance (ANOVA) was used with variability partitioned at two levels; between plots or field units (to compare resources) and within plots (to compare batches and interaction effects). Data were log transformed prior to analysis to help approximate to the assumptions of constant variance and normal distribution of the ANOVA, with log(count + 1) transformation being used for variables that included zero values. Results are presented as geometric means (± SE) for each crop and batch.

7.5.4 Results

Contrary to expectations, we found no difference in maximum size, worker gain or male production between colonies in different AES treatments or cropping regimes (resource-rich environment, oilseed rape; resource-poor environment, wheat). Indeed population and growth parameters estimated from the experimental *Bombus terrestris* colonies showed that any positive effects of increased food resources (determined by crop abundance not margin options) were offset by an increased likelihood of invasion by social parasites (Figure 7.14). Colonies founded earlier in the year were significantly more likely to be parasitized by cuckoo bees than those founded later in the year (ANOVA, $F_{1,10}$ =11.35, P=0.007). In addition colonies next to highly abundant food resources such as oilseed rape were significantly more likely to be parasitized by cuckoo bees than colonies next to sparser food associated with wheat fields (ANOVA, $F_{1,10}$ =19.5, P=0.001).

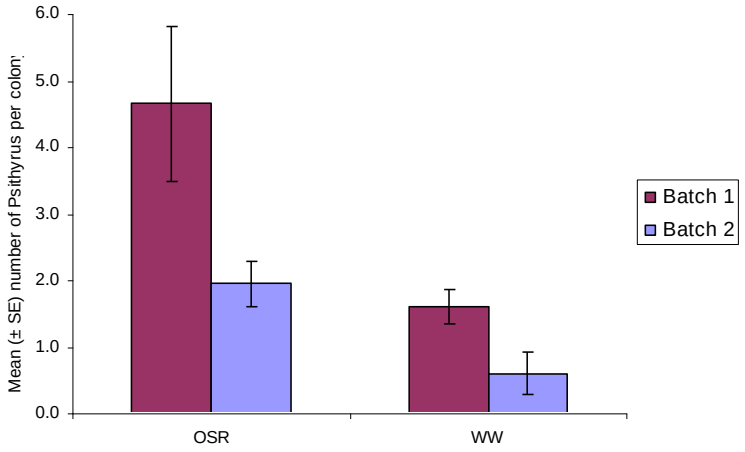


Figure 7.15 Differences in intensity of parasitism by *B.* (*Psithyrus*) *vestalis* for *Bombus terrestris* colonies surrounded by oilseed rape (OSR) and wheat fields (WW). Colonies in Batch 1 (crimson bars) were placed in the field in early May, those in Batch 2 (blue bars) in early June.

7.5.5 Discussion

The results suggest that a much higher frequency and intensity of parasitism by the cuckoo bumblebee, *B. (Psithyrus) vestalis*, in colonies within oilseed rape fields offset any positive impacts of resources on population growth parameters. This study therefore provides the first empirical evidence that proximity to abundant forage resources and early nest founding can increase the probability of social parasite invasion and thus offset the potential positive effects of these factors on bumblebee colony performance.

Bumblebee colonies show considerable variation in life history characteristics such as growth rate and reproductive output, both within and between species (Müller &

Schmid-Hempel, 1992; Goulson *et al.*, 2002; Ings *et al.*, 2006). Nevertheless our findings contrast with other studies investigating the influence of environmental factors on colony performance. Pelletier and McNeil (2003) showed that worker number and reproductive success were increased by supplementing food resources inside field colonies of *B. impatiens* and *B. ternarius*, but that this supplementation had no effect on the probability of successful *Psithyrus* invasion. Our study suggests that where natural food availability is enhanced in the surrounding landscape, *Psithyrus* females are likely to be more active and hence the probability of a colony being invaded increases. The frequency of invasion was extremely high, reaching 92% amongst fields of oilseed rape. This compares with variable rates of invasion elsewhere, from between 20-40% of *B. lapidarius* colonies observed in England (Sladen, 1912) and 30% of *B. lucorum* colonies in Switzerland (Müller & Schmid-Hempel, 1992) to almost 100% of *B. impatiens* colonies in one year in Canada (Pelletier & McNeil, 2003; see also Alford, 1975).

Several factors could account for the observed behaviour of cuckoo bumblebees in our experiment. Following emergence from hibernation, *Psithyrus* females require both pollen and nectar to develop their ovaries and gain energy for flight, and they have been recorded foraging frequently in oilseed rape (Haughton *et al.*, 2003). The high density and super-abundance of forage in such crops may be particularly attractive to *Psithyrus* which are generally weaker in flight than *Bombus* queens (Alford, 1975), and may be less able to exploit sparsely distributed resources.

In terms of the effect of colony founding time on both invasion and usurpation, it seems probable that the later batch of colonies were less susceptible because most nest-searching *Psithyrus* and *B. terrestris* queens had, by that time, already found a host (Müller & Schmid-Hempel, 1992). There are few data on the frequency of nest usurpation attempts by *Bombus*, but it is likely to be more prevalent where and when natural nest sites are rare. The high proportion of 'early-founded' colonies in which usurping *B. terrestris* were found (63%) suggests that suitable nest sites may have been limited in our agricultural landscape relative to the abundance of *B. terrestris* queens. This highlights the need for a greater emphasis on bumblebee studies focused at the colony and population-level, given the difficulty of assessing effects of natural nest site availability on species' survival and persistence.

These findings do not support the hypothesis that indirect effects of food scarcity can increase susceptibility to brood parasitism, as shown previously for bumblebees (Cartar & Dill, 1991) and for the solitary bee, *Osmia pumila*, which experienced a higher probability of brood parasitism in cages with sparse floral resources than in those with rich resources (Goodell, 2003). Our results do, however, agree more closely with other findings from eusocial Hymenopteran systems. Downs and Ratnieks (2000) showed that in honeybees, colonies under food stress were heavily guarded and less tolerant of intruding non-nest mates than when resource availability increased. Similarly, food stress has been shown to increase rejection of larvae of the cuckoo butterfly, *Maculinea rebeli*, by host colonies of some *Myrmica* ant species (Elmes *et al.*, 2004).

In conclusion, our study provides some evidence that resource availability and colony founding date may be important factors in determining the probability of social parasite invasion in bumblebee colonies, in addition to influencing aspects of colony performance such as growth and reproduction. Founding a nest near super-abundant forage resources may enhance colony growth under 'enemy-free' conditions (e.g. Goulson, 2002; Pelletier & McNeil, 2003) but carries a greater risk of invasion by *Psithyrus*, especially early in the season, when usurpation by other *Bombus* queens is also more likely. In contrast, colonies founded amongst sparse or unpredictable resources necessitate longer foraging flights and risk slower rates of growth (Westphal *et al.*, 2006), but reduce the probability of brood parasitism and retain the ability to reproduce. Optimal nest location may therefore be among intermediate levels of resources or where there is high spatial heterogeneity in forage resources. Where resources and parasites are abundant, it may be beneficial for a queen bumblebee to nest later in the season.

These results suggest that natural variations in resource availability and parasitism rates could be partly responsible for the maintenance of variation in colony founding dates within *Bombus* species. In an applied context, our findings support efforts to enhance the quality and quantity of semi-natural habitats for bumblebees within intensively managed landscapes (e.g. agri-environment schemes, Carvell *et al.*, 2007). Enhancing the availability of forage resources where they are scarce and diversifying resources in areas such as those currently dominated by single mass-flowering crops, thus increasing spatial heterogeneity in

resources, providing a number of refuges for bumblebees and perhaps reducing the effects of social parasite invasion.

7.6 **Moths**

Lepidoptera are one of the four largest orders of insects (New 2004a) with around 2,500 species in the UK (Waring *et al.* 2003). They are extremely important food sources (especially their larvae) for many other taxa such as bats and birds (Wilson *et al.* 1999) and thus a vital component of most healthy ecosystems. For example, the estimated number of Lepidoptera larvae eaten by Blue Tit, *Cyanistes caeruleus*, chicks in Britain each year is around 35 billion (Fox *et al.* 2006). Within Lepidoptera, butterflies (or the 'Rhopalocera') have been particularly well studied such that we have good information on individual species ecology, distribution and population abundance throughout Britain (e.g. Botham *et al.* 2010; Fox *et al.* 2011). Moths however, are much less well researched and whilst the ecology and distribution of a number of species is relatively well known, for many it remains poorly understood. However, moths are much more diverse and account for most of the Lepidopteran biomass with only 59 resident species of butterfly in the UK, the rest being moths. As such they deserve greater consideration with regards to how environmental change brought about by climate and anthropogenic factors will impact their abundance and distribution and the knock-on effects on the communities in which they are important components (New 2004).

Like butterflies, moths are regarded as good indicators of environmental change, showing high sensitivity to changes in the climate and habitat management for example (Fox *et al*. 2006). Recent publications have shown severe declines in a large number of Britain's larger "macro" moth species (Conrad *et al.* 2006; Fox *et al.* 2006). These declines are particularly pronounced in arable habitats (Conrad *et al*. 2006) with a number of 'common and widespread' species given Biodiversity Action Plan Priority species status (Thomas *et al.* 2006). Farmland is also a major habitat for a number of scarcer species, such as the Pale Shining Brown *Polia bombycina*, which is now only found in a limited number of colonies on farmland in southern England(Merckx *et al.* 2010b).

More recently, the importance of moths has become better recognised with a number of national schemes and events aimed at quantifying their abundance and distribution. In parallel, moths have been included in studies assessing the effectiveness of

AES. These studies have focussed on comparing pairs of sites across regions of the UK and have found little evidence that any particular subscription has benefitted all moths in a similar way. Work by Merckx *et al.* (2009a; 2009b; 2010a) has shown that hedgerow trees are more important for larger moths than widened field margins under AES, but that this is only significant where there are widened field margins suggesting a need for subsidies to encourage the management of both (Merckx *et al.* 2009b). In addition though, these benefits were only found to be significant for species of moth that are less mobile and the underlying opinion is that AES need a landscape-scale approach to be effective for a wider range of species (Merckx *et al.* 2010a) In contrast, Fuentes-Montemayor *et al.* (2011) found that moth abundance and diversity was generally greater within field margins and species-rich grasslands under AES management compared to conventionally managed farmland habitats and that increasing the cover of semi-natural habitat at a local scale does benefit a wide range of moths. The increase in abundance and diversity, however, was not significant for macro-moths, but was for micro-moths, a group of moths arbitrarily split historically on the basis of their size and ignored in other studies (New, 2004) due to difficulties in identification and their huge diversity (over 2,400 species of moth have been recorded in the UK of which less than 850 are 'macro-moths' (Young 1997; Waring *et al.* 2003)). None of these studies has looked at the temporal patterns in moth abundance and richness on farms under AES. Thus they do not address one of the major criticisms of studies into the effectiveness of AES – testing against a baseline (Kleijn & Sutherland 2003).

7.6.1 Objectives

Our overall objective was to assess how the abundance and diversity of moths were affected by AES at the whole farm and treatment scales over 5 years. Further we wanted to understand if there were differential impacts on groups of moths (micro- *vs.* macro) and establish the relative effects of both treatment and landscape (e.g. habitat context) scales on abundance.

7.6.2 Methods

Moth abundance was recorded across Hillesden farm using Robinson style light traps fitted with 125W MV lamps. Traps were run overnight using petrol generators. These traps work using the 'lobster-pot' principle. Moths are attracted to the light source which is positioned centrally within a cone at the top of the trap. The moths then enter the cone into a large

circular container from which they find it difficult to exit. Within each container egg cartons are placed to provide refuge for captured moths. The following morning after a trap has been deployed the traps are sorted with number and species identity recorded. From 2008 onwards both micro- and macro-moth species were recorded to species level using a combination of in situ identification and post-collection identification by Peter Hall, a micro-moth specialist, who examined the genitalia of specimens. Before 2008 micro-moths were not recorded to species level, but their abundance as a group was recorded.

Moths were trapped twice a year (late-May and late-July) in all years except 2006 where they were trapped only once (late-July). These are widely regarded as the peak times of year for moth abundance and diversity. In each period the week chosen to trap was when moonlight was lowest (3-4 days either side of the New moon) to minimise moonlight effects and to standardise this for each trapping session. In each trapping period one Block was surveyed each night, with one moth trap for each treatment within that block. In 2006 and the first period of trapping in 2007 there were three treatments: Cross-compliance (CC); Entry Level Scheme (ELS) – 'tussocky margins'; and Entry Level Scheme Extra (ELSX) – margins sown with wildflower mixes. For CC treatment traps were placed on the narrow grassy strip between the hedgerow and crop, whilst in other treatments traps were placed in the centre of the field margin that had been sown with the prescribed treatment. All traps were placed within 2m of the hedgerow boundary of the field. Subsequent to these initial trapping sessions a fourth treatment was added as a local control for the ELSX from the second period of 2007 onwards (ELSX NM). This included placing a trap in a field that had an ELSX margin but where the trap was placed on the opposite side of the field where there was no margin c.f. the CC treatment. The order in which treatments were sampled was randomised each year and each trapping period. Whilst we recorded all moths on the outside and inside of the trap including a 1m radius search in the vegetation surrounding the trap, only moths recorded from the inside of traps were used in analysis.

7.6.3 Species

Total abundances and α-diversity (species richness) were recorded from each trap as the response variables for analysis. A number of trap data were excluded due to electrical failures and where trap counts were extremely low because of local climatic conditions. For example, there were occasions where one trap of the four would catch very few moths

because it was more exposed on a cold night. This is unlikely to be due to treatment effects and most likely to do with its position on the farm, the climatic conditions for which we do not have accurate measurements to account for. Thus, traps with less than 20 individuals and/or less than 10 different species were excluded from the analysis. Moths were split into macro- and micro-moths. These groups vary in their characteristics and functional traits (e.g. body size and dispersal capabilities) and have demonstrated different responses to agri-environment schemes (Fuentes-Montemayor *et al*. 2011). Macro-moths were further broken down into broad feeding groups based on their main host plant types to see whether different types of moth showed different responses to ELS. These were: Grass feeders (species feeding predominantly on grasses, rushes, sedges), Herb feeders (species feeding predominantly on herbaceous plants), Lichen feeders and Woody plant feeders (species feeding predominantly on trees and/or shrubs). Finally, we looked at Section 41 Priority species as a separate group. A large number of macro moths have recently been listed as BAP Priority (many for research only) species in the UK due to significant declines in their population (Conrad *et al*. 2006) and distribution (Fox *et al*. 2006). Farmland moths in particular have shown significant declines, and many common and widespread species have now become much less abundant even though they may still be found over a wide range. In addition, some of the rarer species on this list, such as the Pale Shining Brown, *Polia bombycina*, are now only found on farmland, and research has shown that successful farmland management can benefit such species (Merckx *et al.*, 2010b). This group in particular, therefore, is one which agri-environment schemes should aim to benefit.

7.6.4 Weather

Weather affects the effectiveness of moth traps greatly (McGeachie 1989; Yela & Holyoak 1997). 2007 and 2008 were poor years for flying insects throughout the UK (e.g. Botham *et al.* 2008; 2009; 2010), due to the heavy rainfall and lower than average temperatures experienced for a large proportion of the spring and summer months during which they normally fly. Thus, we included climatic variables in our analysis to account for this. Monthly temperature data were obtained from the Hadley Central England temperature (HadCET) series available from the Meteorological Office (Parker *et al.* 1992). Monthly precipitation data were obtained from the England and Wales Precipitation (EWP) series, also available from the Meteorological Office (Alexander & Jones 2001). From these the monthly data

corresponding to the trapping periods (May and July) for each year were extracted and used as the weather variables for analysis.

7.6.5 Statistical analyses

Analyses were broken into 3 main components:

(i) Testing overall moth abundance and diversity at the farm-scale after 5-years of ELS in 2010 against the baseline in 2006

(ii) Testing for treatment differences in moth abundance and diversity at the farm-scale over the 5-year period (2006-2010)

(iii) Testing for habitat effects on moth abundance and diversity at the landscape-scale

All data were count data and thus followed a Poisson distribution. For (i) we used two approaches. Firstly we analysed the data using a Mann-Whitney U-test to test whether abundance and/or diversity had significantly changed in 2010 from 2006 levels. We then performed more detailed analysis to incorporate the effects of weather on moth trap catches and to determine more detailed patterns in how moth abundance and diversity changed over the 5-years. This was achieved by using generalised linear models with a quasi-Poisson error distribution (to account for over-dispersion in the data) and a log link function. Temperature and precipitation data from the month during which the traps were operated were entered into the model along with year and its quadratic term (prior graphical analysis showed that the relationship between moth abundance and diversity over time was not a linear relationship):

n or α = year + year^2+temperature+precipitation

For (ii) we used a similar model with the addition of treatment:

n or α = year + year^2+temperature+precipitation + treatment

And for (iii) we used a similar model again but with the different habitat variables included:

n or α = year + year^2+temperature+precipitation + Arable crop + Grass habitat + Forb-rich habitat + Woody plant volume.

In each part (i)-(iii) we performed the models separately for macro- and micro-moths and also for Section 41 species of macro-moths. For (iii) we also ran the habitat models for different groups of macro-moths defined by their host plant type as described earlier (Grass-feeders, Herb-feeders, Lichen-feeders and Woody-plant feeders). In addition, in part (iii) we

repeated each model at four different habitat scales (100m, 250m, 500m, and 1km). We performed backwards, stepwise regression using F tests to remove any non significant term and to define the minimum adequate model. All statistical analyses were carried out using R version 2.12.0 (R development core team, 2009) whilst graphical analyses were performed using a combination of Minitab version 14 and R version 2.12.0.

7.6.6 Results

7.6.6.1 Overall moth abundance and diversity at the farm-scale after 5-years of ELS

Initial analysis using Mann-Whitney U tests to test for differences in abundance and diversity between 2006 and 2010 showed that both abundance and species richness of macro- and abundance of micro-moths was significantly lower at the end of the ELS in 2010 compared to the start of ELS in 2006 (Macro-moths: abundance, $W = 443$, $P = 0.007$; α-diversity, $W = 467$, $P = 0.001$; Micro-moths: abundance, $W = 430$, $P < 0.001$; see Figure 7.16 and Table 5.1). Conversely, Section 41 species abundance and diversity was significantly greater in 2010 than in 2006 (abundance, $W = 99$, $P < 0.001$; α-diversity, $W = 94$, $P < 0.001$; see Figure 7.16).

However, these simplistic analyses fail to incorporate variation in weather, which greatly affects the nightly catches in moth traps, and natural annual fluctuations in Lepidoptera populations. 2007-09 were particularly poor years for Lepidoptera, with unfavourable climatic conditions during a high proportion of the months during which we trapped and during which the greatest number of Lepidoptera normally fly (Botham *et al*, 2009). Poisson regression models showed that macro-moth abundance significantly changed since 2006 in a non-linear relationship (Figure 7.17) such that abundance started high in 2006 and then dropped dramatically in 2007 and 2008 increasing again thereafter to a level slightly lower in 2010 than in 2006. Micro-moth abundance showed the same pattern, but the abundance of UK BAP Priority species showed the opposite pattern, with highest abundance in the middle years. Further investigation showed that this was because of two species, Small Square-spot, *Diarsia rubi,* and Large Nutmeg, *Apamea anceps,* which were particularly abundant in 2008 and 2009. This pattern also meant that there was a significantly negative association between Section 41 species abundance and temperature

as these middle years were generally cooler. Precipitation also had a significantly negative effect on Section 41 species abundance.

Diversity did not significantly change through time for macro-moths, though after accounting for weather effects it was greatest in 2010 (Figure 7.17). Temperature had a significantly positive effect on macro-moth diversity. We did not have data on the diversity of micro-moths for 2006 and 2007, but there was a significant change in the last three years of ELS whereby diversity increased and was greatest in 2010 (Figure 7.17). Micro-moth diversity (in this latter period) was significantly affected by precipitation, with greater rainfall during trapping sessions associated with a greater number of micro-moth species. Section 41 species diversity also significantly changed since 2006, showing the same non-linear pattern as for macro-moths generally. Temperature had a significantly negative effect on Section 41 diversity.

Table 7.1. Summary of Poisson regression model outputs for annual abundance and α-diversity of different moth groups and the effects of weather. Direction of effect is given using (-) negative, and (+) positive. Empty cells refer to non-significant terms. n/a= analyses could not be performed because not all years available.

	Significant change since baseline		**Significant effect of Temperature ^{o}C**		**Significant effect of Precipitation (mm)**	
	Abundance	α-diversity	Abundance	α-diversity	Abundance	α-diversity
Macro-moths	-			+		
Micro-moths	-	n/a	+	n/a		n/a
Section 41 species	+	+	-	-	-	

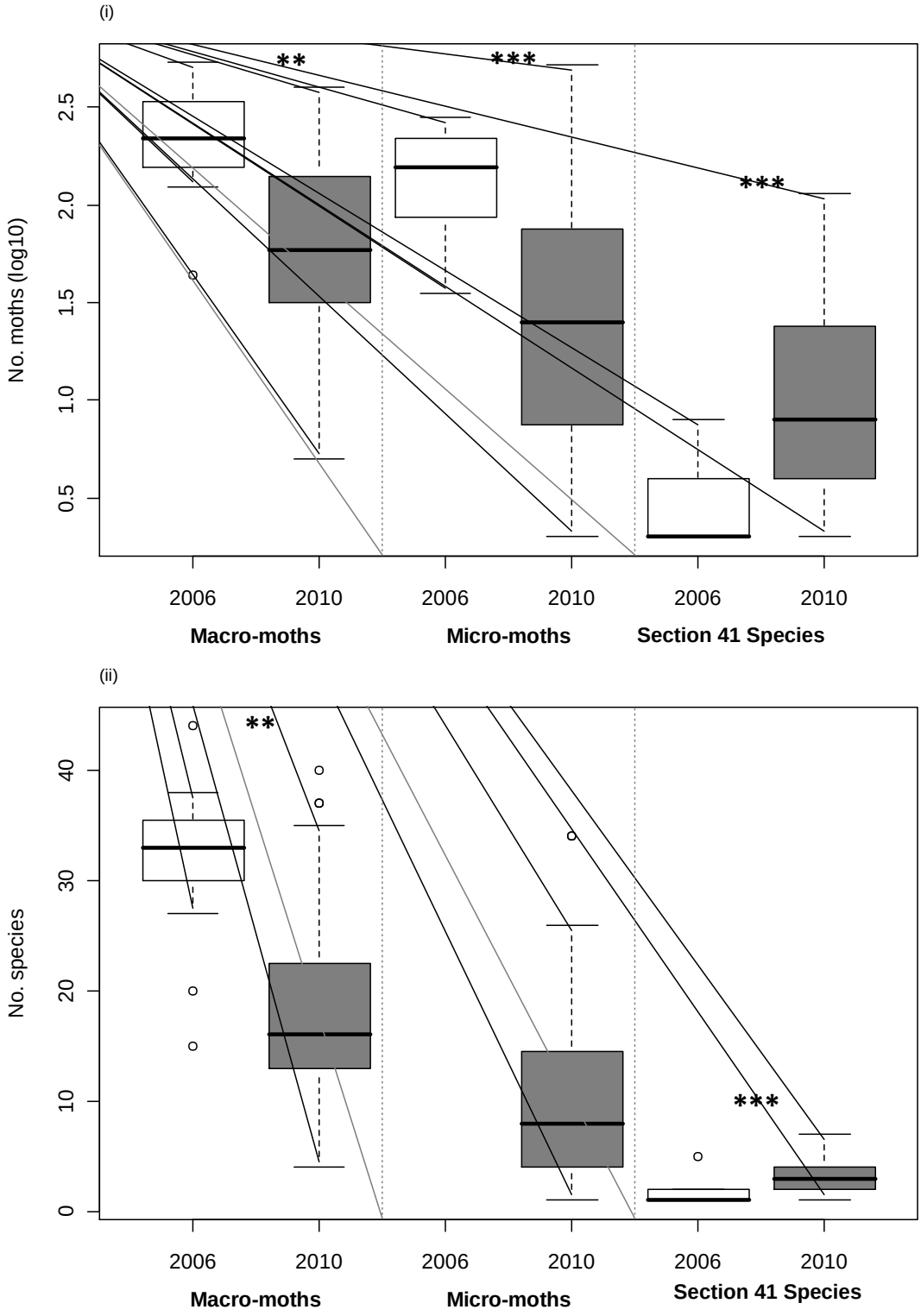


Figure 7.16. Median (with interquartile range and range) abundance (i) and α-diversity (ii) of different moth groups at the start (2006) and end (2010) of the 5-year period under ELS. Note: α-diversity was not calculated for Micro-moths in 2006. **P < 0.01, ***P < 0.001 (Mann-Whitney U test).

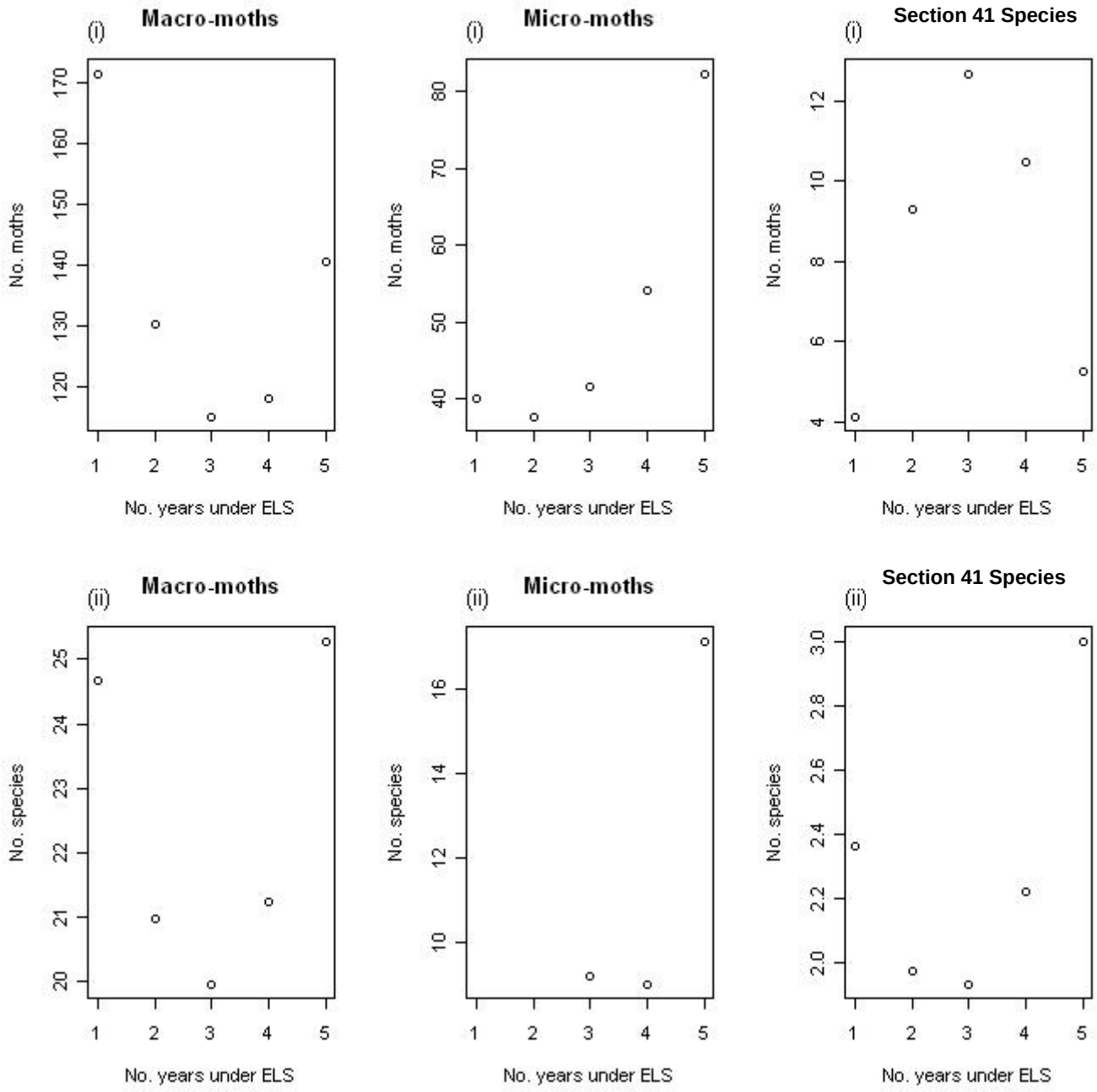


Figure 7.17. Plots to show the annual abundance (i), and α-diversity (ii) of different moth groups against the number of years Hillesden was under ELS (1-5 = 2006-2010). These plots show values calculated from the coefficients generated by Poisson regression models with weather accounted for and are based on a constant value for temperature and precipitation which is based on the mean of the all trapping data for the 5-year period: temperature (14.7^{o}C) and precipitation (74.9mm). Note: data was not analysed for micro-moth α-diversity in the first two years of ELS (2006 and 2007).

7.6.6.2 Treatment differences in moth abundance and diversity at the farm-scale

Preliminary analysis showed that there was no significant difference across treatments in either abundance or α-diversity of macro-moths (Kruskal-Wallis: abundance - χ^2 = 1.54, d.f = 3, P = 0.672; α-diversity - χ^2 =1.49, d.f = 3, P = 0.683). The same was true for Section 41 species (Kruskal-Wallis: abundance - χ^2 = 3.62, d.f = 3, P = 0.306; α-diversity - χ^2 =1.78, d.f = 3, P = 0.619). However, whilst there was also no significant difference in α-diversity across treatments (Kruskal-Wallis: χ^2 =2.71, d.f = 3, P = 0.439), the abundance of micro-moths significantly differed between treatments over the 5-years of ELS (Kruskal-Wallis: χ^2 =9.42, d.f = 3, P = 0.024). Post-hoc analysis using Mann-Whitney U tests showed that micro-moth abundance was significantly greater in the ELSX treatment than both the CC (W = 418, P = 0.015) and ELSX NM (W = 703, P = 0.007) treatments (see Figure 7.18).

More detailed analysis using Poisson regression models supported these results (Table 5.1). In addition, we found a significant difference in the diversity of micro-moths between CC and ELSX treatments, showing ELSX to have significantly greater diversity regardless of year and weather. Although treatment rarely had a significant effect, there is a visible trend in the data towards greater abundance of macro- and micro-moths in the ELSX treatment than all other treatments, with ELS also supporting more macro- and micro-moths than CC and ELSX NM treatments (Figure 7.18). Section 41 species abundance differed less between treatments, probably because the main species (*D. rubi* and *A. anceps*) contributing to their abundance are generalist grass feeders and were abundant in all treatments. ELSX also supported greater diversity for all moths, but the differences between other treatments were much more variable and ELS did not always support more species than CC and/or ELSX NM (Figure 7.18).

Table 7.2. Summary of treatment effects for macro-, micro- and Section 41 moths from Poisson regression model which also included weather and year variables (not described here). Statistical significance of these effects are given: **P < 0.01, *** P < 0.001.

Treatment	Response	Macro-moths	Micro-moths	Section 41-moths
ELS vs CC	Abundance	ELS>CC	ELS>CC	ELS<CC
	α-diversity	ELS>CC	ELS>CC	ELS<CC
ELSX vs CC	Abundance	ELSX>CC	**ELSX>CC*** **	ELSX>CC
	α-diversity	ELSX>CC	**ELSX>CC***	ELSX>CC
ELSX NM vs CC	Abundance	ELSX NM>CC	ELSX NM<CC	ELSX NM>CC
	α-diversity	ELSX NM>CC	ELSX NM>CC	ELSX NM>CC
ELSX vs ELS	Abundance	ELSX>ELS	**ELSX>ELS** **	ELSX>ELS
	α-diversity	ELSX>ELS	ELSX>ELS	ELSX>ELS
ELSX NM vs ELS	Abundance	ELSX NM>ELS	ELSX NM<ELS	ELSX NM>ELS
	α-diversity	ELSX NM<ELS	ELSX NM<ELS	ELSX NM>ELS
ELSX NM vs ELSX	Abundance	ELSX>ELSX NM	**ELSX>ELSX NM*** **	ELSX>ELSX NM
	α-diversity	ELSX>ELSX NM	ELSX>ELSX NM	ELSX<ELSX NM

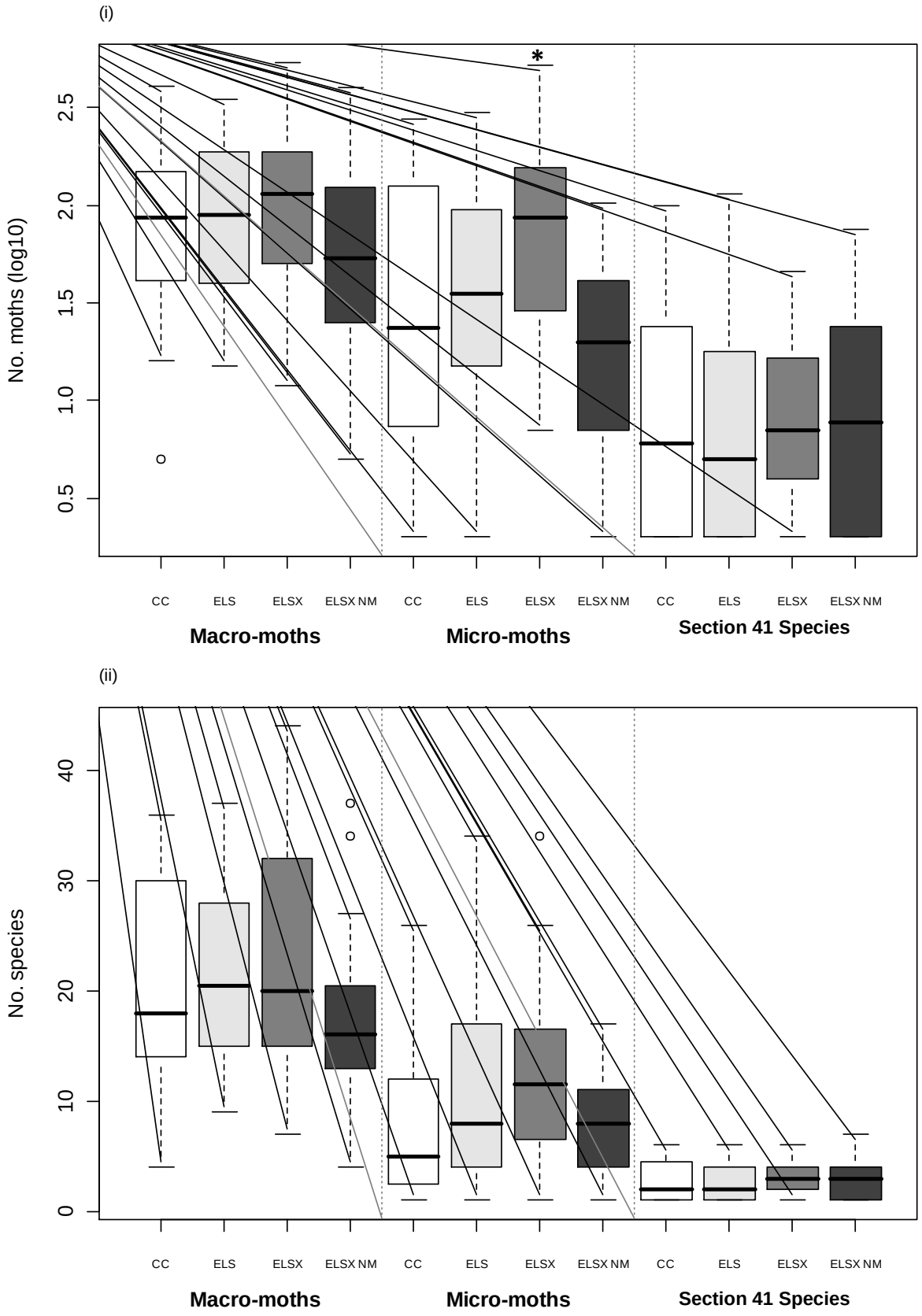


Figure 7.18. Median (with interquartile range and range) abundance (i) and α-diversity (ii) of different moth groups under different treatment prescriptions over the 5-year period under ELS. *P < 0.05 (Mann-Whitney U-test: ELSX significantly greater than CC and ELSX NM).

7.6.6.3 Habitat effects on moth abundance and diversity at different spatial scales

Before analysing models to investigate the effects of different habitat types on moth abundance and diversity we first quantified the different amount of each habitat type around each treatment type at different spatial scales

Figure 5.1). Table 7.3 shows a summary of linear regression analysis on the amount of different habitat types around moth traps in each treatment. As one would expect, at the local scale (100m), moth traps positioned in CC treatments were surrounded by a significantly greater amount of crop than any other treatment, with traps in ELSX having significantly less crop surrounding them than any other treatment. This pattern changed with scale and at 500m and 1km traps in CC treatments were actually surrounded by significantly less crop than other treatments and the most crop was found around traps in ELS treatments. Grassy habitat was greatest around traps in ELS treatments at the local scale, and least around traps in CC treatments, but this completely reversed at the 1km scale. The least amount of forb-rich habitat was also found around traps positioned in CC treatments, and, again not surprisingly, forb-rich habitat was greatest around ELSX traps at the local scale. This was true at most scales except the 1km scale, where again there was a reversal. Finally, the amount of woody plant cover was greatest around traps located in CC treatments, significantly more than around traps in both ELS and ELSX NM treatments but not the ELSX treatment at the 100m scale. Unlike for other habitat types, this pattern held true at all most scales including the 1km scale. For more detail on between-treatment differences at each scale refer to Table 7.3.

Table 7.3. Differences between treatments (CC, ELS, ELSX, ELSX NM) in amount of different habitat types (Crop, Grass, Forb-rich, Woody plant) around moth traps at different spatial scales (100m, 250m, 500m, 1km). In each cell the treatment listed is that which has the highest amount of the given habitat type between the pair of treatments listed in the column heading. Columns 9 and 10 show the treatment that has the least and most respectively of a given habitat type at a given spatial scale. Statistical significance is given: *P < 0.05, **P < 0.01, ***P < 0.001.

Habitat type	Scale	CC v ELS	CC v ELSX	CC v ELSX NM	ELS v ELSX	ELS v ELSX NM	ELSX v ELSX NM	Treatment with least	Treatment with most
Crop	100m	CC***	CC***	CC***	ELS***	ELS*	ELSX NM***	**ELSX**	**CC**
	250m	ELS	CC***	CC**	ELS***	ELS***	ELSX NM**	**ELSX**	**ELS**
	500m	ELS**	CC.	ELSX NM	ELS***	ELS	ELSX NM**	**ELSX**	**ELS**
	1km	ELS***	ELSX***	ELSX NM***	ELS*	ELS	ELSX NM*	**CC**	**ELS**
Grass	100m	ELS***	ELSX*	ELSX NM***	ELS***	ELS.	ELSX NM**	**CC**	**ELS**
	250m	ELS	CC	ELSX NM*	ELS	ELSX NM	ELSX NM*	**ELSX**	**ELSX NM**
	500m	CC	CC	CC.	ELSX	ELS	ELSX	**ELSX NM**	**CC**
	1km	CC***	CC**	CC***	ELSX.	ELSX NM	ELSX	**ELS**	**CC**
Forb-rich	100m	ELS***	ELSX***	ELSX NM***	ELSX***	ELSX NM***	ELSX***	**CC**	**ELSX**
	250m	CC	ELSX***	ELSX NM***	ELSX***	ELSX NM***	ELSX***	**ELS**	**ELSX**
	500m	ELS	ELSX***	ELSX NM***	ELSX***	ELSX NM***	ELSX**	**CC**	**ELSX**
	1km	CC	CC***	CC**	ELS**	ELS*	ELSX NM	**ELSX**	**CC**
Woody plants	100m	CC***	CC	CC***	ELSX**	ELS.	ELSX***	**ELSX NM**	**CC**
	250m	CC***	ELSX	CC***	ELSX***	ELS	ELSX***	**ELSX NM**	**ELSX**
	500m	CC.	CC	CC***	ELSX	ELS*	ELSX***	**ELSX NM**	**CC**
	1km	CC	CC***	CC***	ELS*	ELS*	ELSX	**ELSX NM**	**CC**

Poisson regression outputs from models to investigate which habitat types are important for different moth groups and moth feeding groups are given in Table 7.4. Here we will only summarise the main findings.

7.6.6.4 Effect of habitat on different moth groups

At the local scale (100m) only crop had a significant effect on any moth group, showing a significantly negative association with micro-moth abundance. At the 250m and 500m spatial scales no habitat types were significantly associated with either abundance or diversity of any moth group, but at the landscape scale (1km) we found significantly negative effects of woody plant cover on both the abundance and diversity of macro-moths and on the diversity of micro-moths. The amount of crop within a 1km area also had a significantly negative effect on the diversity of micro-moths.

7.6.6.5 Effect of habitat on different feeding groups (macro moths only)

Macro-moths were further split into broad feeding groups to see if their abundance and diversity was more directly related to the availability of the correct habitat. At the local scale there was a significantly positive relationship between the abundance of grass feeders and the amount of grassy habitat nearby. Whilst this relationship existed at all scales it was only significant at 100m. At larger scales (500m and 1km) the amount of woody plant cover had a significantly negative effect on the abundance and diversity of grass feeders. Species feeding on herbaceous plants showed greater abundances where the amount of forb-rich habitat was greatest, this relationship being significant at the local scales (100m and 250m). The diversity of herb feeders showed a significantly negative relationship with the amount of crop cover at the 100m scale. As for the grass feeders, the abundance and diversity of herb feeders was also significantly negatively affected by the amount of woody plant cover at the landscape scale (1km). The abundance and diversity of lichen feeders was not significantly affected by any habitat type at any scale, but was positively associated with woody plant cover at the local scale as one might expect. Similarly, whilst not significant, the abundance and diversity of woody plant feeders was positively associated with woody plant cover at most scales (though not at the landscape scale where the relationship became negative).

Table 7.4. Summary of the relationship between the abundance and diversity of different moth groups (a) and macro-moth feeding groups (b) and the amount of different habitat types at different spatial scales (100m, 250m, 500m, 1km). The direction of each relationship is given as negative (-) or positive (+) and statistical significance of each relationship is given: *P < 0.05, **P < 0.01, ***P < 0.001. '.' Denotes border-line significance where P is close to 0.05. Note: the direction of non-significant effects are derived from maximum models prior to backwards stepwise regression analysis. Significant effects are those from the final minimum adequate models

Habitat	**Crop**				**Grassy**				**Forb-rich**				**Woody**			
Buffer size (m)	**100**	**250**	**500**	**1000**	**100**	**250**	**500**	**1000**	**100**	**250**	**500**	**1000**	**100**	**250**	**500**	**1000**
(a) Moth group																
Abundance(n)																
Macro	+	+	-	-	+	+	-	+	+	+	+	+	-	-	-.	-**
Micro	-*	+	-	-	+	+	-	-	+	+	-	-	+	+	-	-.
Section 41	+	+	+	+	+	+	+	+	+	+	+	+	-	-	-	-
α-diversity																
Macro	+	+	-	-	+	+	-	-	+.	+	+	+	+	-	-	-*
Micro	-	+	-	-*	+	+	-	-	+	+	-	-	-	-	-.	-**
Section 41	-	-	-	-	+	+	-	-	+	+	+	+	-	-	-	-
(b) Feeding group																
Abundance(n)																
Grass	+	+	-	-	+*	+	+	+	+	+.	+	+	-	-	-**	-**
Herb	+	-	-	-	+	+	-	+	+*	+*	+	+	-	-	-.	-**
Lichen	+	-	+	-	+	-	-	-	+	+	+	-	+	+	+	+
Woody	+	+	+	-	+	-	-*	-*	+	+	+	-	+	+	+	-
α-diversity																
Grass	+	+	-	-	+	+	-	-	+.	+	+	+	+	-	-*	-**
Herb	-*	-	-	-	+	+	-	-	+	+.	+	+	-	-	-	-*
Lichen	+	+	+	-	+	+	+	-	+	+	+	+	+	+	+	-
Woody	-	+	-	-	-	-	-	-	+	+	+	+	+.	+	-	-

The amount of grassy habitat had a significantly negative effect on the abundance of woody plant feeders at the 500m and 1km scales.

7.6.7 Discussion

Generally, the abundance and diversity of moths on Hillesden farm has benefited from the farm being under ELS. Whilst baseline levels show a decline in 2010 compared to 2006 for macro- and micro-moths, more detailed analysis shows that poor weather played a major role in producing these trends. Weather is likely to have directly affected moth catches (McGeachie 1989; Holyoak *et al.* 1997; Yela & Holyoak 1997), as accounted for in our analysis, but also indirectly through negative effects on moth populations for subsequent years. The relationship was non-linear, with steep declines in both abundance and diversity in the initial 2-3 years of ELS followed by increase to levels near to or above those in 2010 depending on the moth group. Indeed, 2007-2009 were regarded countrywide as poor years for Lepidoptera, whilst 2006, the baseline for this study was the last in a run of particularly good years (Botham *et al*, 2007, 2008). Despite such declines there was still a significant increase in the abundance and diversity of Section 41 priority species, a group of macro-moths which have declined significantly across the UK in recent decades, especially in agricultural habitats (Fox *et al*, 2006; Conrad *et al*, 2006). Declining species were also found to benefit from AES treatments on farmlands across Scotland (Funetes-Mayor *et al*, 2011) and studies on farms in southern England showed that certain AES treatments benefit rare species that are becoming increasingly restricted in range (Merckx *et al*, 2010b). A major criticism of many data from which analyses on trends in insect populations are performed is the length of the dataset used. Most insects have cyclic populations through density dependent factors such as parasitism and food availability, and density independent factors such as climate. Thus, long-term data are required to elucidate genuine trends in their populations whereby natural annual fluctuations are accounted for. This is clearly illustrated in the current study, and shows how short-term data can be extremely misleading depending on the start and end points of the data collection. If Hillesden had started ELS a year later then the abundance and diversity of moths would almost certainly have shown a significant increase over the 5 years, a trend already visible in Figure 7.17.

7.6.7.1 Treatment effects

Despite the relatively small sample sizes and great variation between years and blocks, there was a general trend for greater abundance and diversity of moths in the ELSX treatment. In addition, both control treatments, CC and ELSX NM, tended to support the lowest number and diversity of moths. Whilst similar to the CC treatment, the ELSX NM treatment still benefited from having the ELSX treatment within the same field as it generally supported more moths. However, these differences were only significant for micro-moths, where abundance was significantly greater in the ELSX treatment than all other treatments. Micro-moth diversity was also greatest in the ELSX treatment compared to the CC control. A likely explanation for this difference is that a large number of the micro-moths recorded during this study feed in the flower heads of flowering plants such as scabiouses (eg *Knautia arvensis* and *Scabiosa columbaria*) and knapweeds (*Centaurea* spp) which are abundant in the flowery margins under ELSX treatments but largely absent from other margins. For example, several species of the tortricid family Cochylidae were found in great abundance in the ELSX treatments, particularly *Aethes smeathmaniana* and *Cochylimorpha straminea* which both feed on knapweeds (Bradley *et al.* 1979). Although non-significant, macro-moth diversity was also greater in the ELSX treatment, which may also relate to the increased availability and diversity of host plant species, but also to the increased availability of nectar sources since a large number of macro-moths feed at flowers as adults. The ELS treatment showed less of an effect, probably because this is simply a grassy margin and in effect, whilst it increases the quantity of non-crop habitat compared to conventional margins, it does not substantially increase the quality of the habitat and/or diversity of resources. The quality and diversity of habitat rather than quantity is extremely important in increasing moth diversity on small habitat patches (Summerville & Crist 2004).

The absence of significant treatment differences in macro-moth abundance and diversity may be explained by a number of factors. Firstly, we might not necessarily expect any change in abundance between treatments like we might for species diversity. If our hypothesis is that biodiversity generally increases in ELS treatments then we should also expect a greater number of predators, parasitoids, fungi and interspecific competitors. Whilst this would promote the diversity of moth species, it does not necessarily translate to greater overall abundance of moths per se. In fact, more homogeneous habitats typical of

CC treatments may harbour greater abundances of common species which inflate the overall abundance, hence the problem of pest species in arable landscapes (e.g. Andow 1983; Altieri *et al.* 1984). Secondly, macro-moths are generally much larger than micro-moths, and, whilst some species of micro-moths show great dispersal behaviour (eg *Plutella xylostella*; Chapman *et al*, 2002) most macro-moths are likely to have greater dispersal abilities than the majority of micro-moths. Whilst the 'attractiveness' of and range at which different artificial light sources effect moths is still greatly debated it is considered that 125W Mercury-vapour lamps may attract moths from no further than 25m (Baker & Sadovy 1978; McGeachie 1988) and thus only moths flying within a limited distance of the light traps will be attracted and subsequently caught. More mobile species are thus more likely to travel further distances and be caught further afield from their habitats. Merckx *et al* (2009a) found that AES only benefited the less-mobile species and Fuentes-Montemayor *et al* (2011) found that abundance and species richness was only significantly greater in field margins under AES for micro-moths and not for macro-moths. In addition to this, previous studies have used 6W Actinic lamp Heath traps to survey moths whereas we used 125W MV lamps. 125W MV lamps attract considerably greater numbers of moths and are accepted to be attractive over appreciably greater distances than 6W Actinic type lamps (e.g. Robinson & Robinson 1950; Waring 1980; Baker 1985; Young 2005; van Langevelde *et al.* 2011). Thus we might expect the difference in abundance and diversity of the more mobile macro-moths between treatments in the current study to be even less pronounced. This is what we found. Whilst there was a tendency for both abundance and diversity of macro-moths to be greater in the ELSX treatment, there was no statistical significance. This also highlights a caveat of the methodology used. Light-trapping is the most commonly used and efficient way to survey moths. However, it relies on attracting moths to an area and whilst attraction ranges are considered to be relatively small it is not necessarily a sample of the moths in the habitat sampled. Other methods such as night surveys via torchlight (Birkinshaw & Thomas 1999) can and have been used, but generally for single species surveys (e.g. Spalding 1997). Larval counts through sweep-netting, beating and suction-traps could also be used but these are extremely invasive and disruptive, and can often result in damage to the larvae. Closely associated with this, is a third factor which may help explain the lack of difference between treatments for macro-moths. Spatial scale is clearly important when considering how one

measures the benefits of AES (e.g. Ricketts *et al.* 2001; Hinsley *et al.* 2009; 2010; Fuentes-Montemayor *et al.* 2011). Whilst locally, on a small spatial scale (100 – 250m), the habitat surrounding each trap location in the different treatments significantly differed, this was not true at larger scales. For some habitats, traps in CC treatments were actually surrounded by more non-arable habitat at larger spatial scales (>500m – 1km). A number of studies have shown that mobile species regularly travel distances up to 1km, whilst 250m is around the limit for less mobile species (Nieminen *et al.* 1999; Doak 2000; Ricketts *et al.* 2001; Summerville & Crist 2004; Merckx *et al.* 2009a). Given that habitat differed only significantly between treatments at the smaller spatial scales it is likely that we sampled very similar moth habitats at each trap location for the majority of macro-moth species. Any differences between treatments are likely due to less mobile species that are not only less mobile, but are restricted to the habitat type that characterises the treatments at the local scale. In addition, whilst grassy and forb-rich habitats were less abundant locally around the CC treatment, woody plant cover was significantly greater. Some CC treatments were adjacent to small copses for example, and/or contained mature hedgerow trees. Hedgerow trees have been found to be extremely important as an AES treatment benefiting macro-moths, particularly less mobile species (Merckx *et al.*, 2009a, 2010a). Taking these patterns in habitat into consideration and also that the species pool in agricultural habitats is probably limited in the first instance, this makes it very unlikely to see differences in macro-moth abundance and diversity between different treatments. That there was a trend towards ELSX supporting greater numbers therefore, shows that there is a very real benefit to this type of AES treatment.

7.6.7.2 Habitat effects

The importance of the habitat has to some extent already been discussed above in the context of treatment differences in moth abundance and diversity. Fuentes-Mayor *et al* (2011) showed that the type of habitat on which AES treatments are applied also has a significant effect on how successful these treatments are at enhancing biodiversity. Species-rich grasslands under AES management showed the greatest benefits to moths compared to conventionally managed counterparts. Similarly, the beneficial effects of AES treatments including wider margins and hedgerow trees on macro-moths on farms across southern England were mostly only significant when on targeted farms where the quality of the

habitat is already greater than most standard arable land (Merckx *et al*, 2009b). This presumably relates to the potential biodiversity of the habitat on which AES are implemented. The species pool available on agricultural habitats may be very limited in the first instance and thus there may be limited scope for improvement in moth diversity, a process by which moth communities may become homogenised across agricultural landscapes (Ekroos *et al.* 2010). Unfortunately we do not have data for moth abundance and diversity on Hillesden farmland prior to ELS status, thus we have no idea of what species could be present if the habitat supported all species it is capable of. However, 2006 acts as a suitable baseline as no treatments had been in place for long enough to have any effect at this point. That moth abundance and diversity is responding to the implementation of ELS suggests that either ELS is reversing the declines observed across agricultural habitats in southern England, or that it is increasing the diversity and abundance of moths at Hillesden compared to what it has been.

It is not only the habitat that AES are implemented on that is important. Surrounding habitat type and quality are also likely to significantly affect how well AES treatments will function (e.g. Kuussaari *et al.* 2007a; 2007b). Numerous studies have shown that proximity to good quality habitat promotes the success of habitat restoration for example, in promoting biodiversity (e.g. Knop & Herzog 2007; Knop *et al.* 2007; Woodcock *et al.* 2010). Even at the local scale this may be represented by the positioning of AES treatments within the farmland (e.g. Marshall *et al.* 2006; Hof & Bright 2010) – should one regularly disperse AES treatments throughout the habitat or concentrate them in prescribed locations? Clearly, there is an important issue of spatial scale. Spatial scale has been shown to be important regarding the benefits of AES to moths, in previous studies (eg Merckx *et al*, 2009b; Fuentes-Mayor *et al*, 2011) and in the current study. At larger scales the ELS and ELSX treatments ceased to offer significantly greater coverage of their prescribed habitat type compared to control treatments because of the mosaic in which they are positioned. This was reflected in the significant increase in abundance of the smaller and presumably less dispersive micro-moths in ELSX treatments, whereas there were no significant treatment effects for the larger and more dispersive macro-moths. Associated with this, the value of implementing AES treatments lies not only in whether certain treatments themselves support more diversity, but whether they also promote the diversity of their

surrounding habitats. Hinsley *et al* (2010) showed an increase in the use of intervening habitat between treatment patches by some bird species on Hillesden farm. In the current study, moth abundance generally increased after 2006 but with little treatment effect, suggesting that all of the farmland habitat could be benefiting from the ELS treatments. At the landscape scale having areas of improved habitat through ELS and ELSX treatments provides a more heterogeneous habitat. Habitat heterogeneity has been shown to be extremely important for Lepidoptera (Braganca *et al.* 1998; Oliver *et al.* 2010). So, whilst 5 years of ELS may not have significantly affected the abundance and diversity of all moths at a local scale, it is very likely it may have done at a larger scale by increasing the heterogeneity of an otherwise very homogeneous habitat. To assess this one would need to measure moth diversity in arable habitat surrounding Hillesden that has not been under any AES over the same time periods.

7.6.8 Conclusions

Whilst at a local scale only micro-moths seemed to benefit significantly from ELS, generally there was a greater abundance and diversity for most moth groups in fields with ELSX and ELS treatments compared to conventionally managed fields. For micro-moths, presumably with lower mobility, ELS treatments offer extremely valuable habitat refugia in an otherwise unsuitable habitat. The outstanding result of this study is that the basic ELS treatment itself does not add much biodiversity value, but that the ELSX treatment does and that future AES should aim to provide more forb-rich habitats rather than simple grassy margins. This habitat was associated with greater abundance and diversity of both macro- and micro-moths, significantly so for micro-moths.

At the farm-scale there seemed to be a decline in the abundance and diversity of macro- and micro-moths over the 5 year period. However, when weather was accounted for patterns emerged that showed an initial decline followed by a continued increase to 2010. The unfortunate timing of ELS on Hillesden meant that the beneficial effects of ELS were largely masked by countrywide declines in moth abundance occurring between 2007 and 2009. Despite this, Section 41 priority species were both more abundant and more diverse on Hillesden after 5 years of ELS.

Differences in habitat type are important at a local scale in promoting the diversity of moths, particularly micro-moths, but not so at the landscape scale, where the benefits of

implementing ELS are likely to be in enhancing landscape heterogeneity which is important for Lepidoptera diversity (Benton *et al.* 2003).

This study highlights some of the caveats of the small time-scale at which ELS currently operates: 5 years is insufficient to start observing significant benefits in populations of insects that are often cyclic and greatly affected by climate. In addition, colonisation time is species specific and depends on the surrounding habitat. It may take many species a long time to colonise the newly available suitable habitats created by ELS. In addition, given that the majority of farmland species are likely to be the generalist wider countryside species (which are often more mobile) moth diversity may only be significantly enhanced once some of the less mobile and more specialist species colonise the ELS habitats.

8 Small mammals

In comparison to other some other animal taxa, such as birds and invertebrates, there have been relatively few studies of the effectiveness of agri-environment schemes on mammal populations and diversity (reviewed in Macdonald *et al*., 2007 and Boatman *et al*., 2008). Mammals are important not only for their intrinsic biodiversity value, but also as providers of cultural ecosystem services and the indicators and supporters of other species or groups. In Northern Ireland, Reid *et al*. (2007) found no beneficial effect of the provision of the Environmentally Sensitive Area (ESA) agri-environment scheme on the abundance of Irish Hares (*Lepus timidus hibernicus*), while Browne and Aebischer (2003) reported benefits of the Arable Stewardship scheme for Brown Hares (*Lepus europaeus*) in East Anglia but not the West Midlands (both conservation priority species). Hof and Bright (2010) found that field margins and hedgerows managed under Environmental Stewardship were intensively utilised by radiotracked Hedgehogs (*Erinaceus europaeus*), possibly as a result of greater availability of their invertebrate food, but there was no comparison with a conventional farm setting. Agri-environment schemes may also benefit generalist 'pest' species, such as Red Fox (*Vulpes vulpes*) and European Rabbit (*Oryctolagus cuniculus*) (Reid *et al*., 2007).

Small mammals, such as mice, voles and shrews, are the main prey of flagship predators such as the Barn Owl (*Tyto alba*), and Askew *et al*. (2006) predicted increased foraging habitat for Barn Owls with the introduction of Environmental Stewardship (ES).

Small mammals are the most common subjects for the study of the effectiveness of agri-environment schemes due to their ability to respond rapidly to effects at a variety of spatial scales, and MacDonald *et al*. (2007) suggested the Wood Mouse (*Apodemus sylvaticus*) as a model species in this respect. Field-scale studies have found a positive response by Wood Mice to a reduced spraying regime on conservation headlands (Tew *et al*., 1992) but a mixed response to set-aside related to the context of management and other habitats such as field margins (Tattersall *et al*., 1999a; 1999b). At the farm scale, Tattersall *et al*. (1999a) found that Wood Mice were most abundant on field margins and that this was probably related to hedgerow proximity, while grassy field margins were likely to be important for other species such as Field Vole (Microtus agrestis), Harvest Mouse (*Micromys minutus*), Common Shrew (*Sorex araneus*) and Pygmy Shrew (*Sorex minutus*). Shore *et al*. (2005) demonstrated that grassy margins provided under ES had a greater abundance of Common Shrews and Bank Voles (*Myodes glareolus*) than conventional margins, and that the widest margins and those abutting hedgerows held the greatest numbers of voles, but Wood Mice showed no response to margin type. Total biomass of small mammals also showed an increase on grassy margins between spring and autumn, but not on conventional margins, a result that supported the prediction of Askew *et al*. (2006) in the ability of ES to provide better foraging for Barn Owls.

Macdonald *et al*. (2000) also examined seasonal usage of field margins by Wood Mice, and reported a greater use by individuals during winter relative to the cropped area of the field, and a greater use of hedgerows after the crop was harvested. In addition to providing foraging habitat, field margins and associated hedgerows may also be important for providing connectivity for mammals in arable landscapes, and the landscape context is now considered as a crucial factor in the success of agri-environment schemes in increasing the diversity and abundance of mammal populations (Macdonald *et al*., 2007). Boatman *et al*. (2008) suggested that, although higher level schemes may offer more benefits at a field- or farm-scale, it is entry level schemes that will be more significant overall due to the relative coverage at the landscape scale. However, the evidence base for assessing the benefits of specific treatments and generic effects on mammals remains limited, and the absence of robust monitoring has so far been a limitation in evaluating most comparisons (Macdonald *et al*., 2007; Boatman *et al*., 2008).

This section describes the small mammal experiment on the Hillesden Estate, which was designed to address these limitations by the use of intensive, large-scale, long-term monitoring of small mammal populations in response to agri-environmental treatments. We investigated the effect on relative abundance, species richness, biomass, and over-winter survival of mice, voles and shrews on field margins of three agri-treatments: conventional Cross Compliance (CC), Entry-Level Scheme (ELS) and Entry-Level Scheme Extra (ELSX). ELS and ELSX treatments were established on the sites of former CC margins in five replicate blocks on the estate, with remaining CC margins in each block acting as a control with which to assess treatment effects after installation. Monitoring began in the Autumn of 2005 to coincide with establishment of the ELS and ELSX field margins, creating baseline data for the mammal populations. Trapping took place during November/December of 2005, 2006, 2008 and 2010 (Years 0, 1, 3 and 5) to assess mammal populations at their annual post-breeding peak, with a further trapping period the following Spring (May) to assess populations at their annual minimum and determine over-winter survival.

8.1 **Methods**

Small mammals within each treatment margin in each replicate block were sampled using a pair of spatially-distinct transects, situated on the field margin at a distance of 2 m from the centre of the bordering hedge or ditch. Transects consisted of 11 Longworth live-traps spaced 10 m apart, parallel to the field margin, and were baited with wheat, carrot and casters (blow-fly pupae). Traps were set on pre-bait for four nights in order to encourage animals to frequent them without being trapped, and then set to catch for the next three consecutive nights. Traps were checked each morning and evening, giving five trapping sessions over *c*.72 hours, or 110 trapping opportunities per treatment per replicate.

On first capture in a season animals were recorded to species, along with weight (to 0.5 g), gender and breeding condition. Wood Mice *Apodemus sylvaticus*, Bank Voles *Myodes glareolus* and Field Voles *Microtus agrestis* were implanted with uniquely-coded RFID 'PIT tags' during Autumn, which enabled subsequent identification on recapture for the course of their life. Fur clips, which would moult out between seasons, were used to denote capture and mark all other species, including Common Shrew *Sorex araneus*, Pigmy Shrew *S. minutus*, Water Shrew *Neomys fodiens*, Harvest Mouse *Micromys minutes* and House

Mouse *Mus musculus*. Animals of all species were also fur clipped in the Spring trapping periods to denote capture.

8.2 **Analyses**

Spring and Autumn periods were analysed separately. Data from both transects of a pair were aggregated when calculating the relative abundance of unique animals on each treatment per replicate, and also for derived variables such as biomass and species richness. Preliminary analyses were necessary, however, in order to test that the proportion of recaptured animals per repeat trapping session (4 sessions per site) did not vary significantly between treatments or years, which may bias the results. We tested this using generalized linear models (glm) with binomial errors in R 2.10.0 (R Development Core Team 2009), using the number of new and recaptured animals as the response (a combined variable which R uses to calculate the proportions). Year was included as a four-level factorial variable, and treatment as a three-level factorial variable. There was no significant interaction of treatment and year in the proportion of animals that were recaptured in trapping sessions during Autumn or Spring (non-significant results not shown). Removal of the interaction terms showed a significant effect of treatment ($F_{164,166} = 3.56$, $P = 0.028$) and year ($F_{164,167} = 6.78$, $P < 0.001$) for Autumn, isolating ELS and Year 3 respectively, and for treatment in Spring ($F_{155,157} = 6.69$, $P = 0.001$), isolating CC. The percentage of trapping sessions that encountered a very high capture rate (> 70%), which could be an important limiter on the number of new animals encountered, was only 9% of Autumn trapping sessions and 7% of Spring trapping sessions. This suggested that any differences in the proportion of recaptures in individual years or treatments would have had little effect of preventing captures of new animals, and thus was unlikely to have influenced the results.

8.2.1 Abundance

Differences in the relative abundance of mammals on each treatment type were compared using generalized linear models. A Poisson distribution and a log-link were used due to the explanatory variable being non-negative and predominantly low integer counts of mammals, although models were re-fitted with a quasipoisson adjustment where over-dispersion of variance in the Poisson model was high. Treatment was used as a factor in the models, and year as a continuous variable, with a 'broken stick' approach being adopted where initial investigation of the response variable indicated a segmentation of the

relationship with predictor variables. Analysis began with the saturated model which generated a separate intercept and slope for each treatment. Step-wise model simplification was then undertaken to test the significance of the main effects and interactions until a minimum adequate model was arrived at, which also allowed determination of the statistical significance of each parameter or interaction in the final model. The most pertinent parameter being tested for was the interaction of treatment over time, which would indicate if the trend in the mean number of animals differed on the three treatments. Non-significant results are not shown.

8.2.2 Species richness

Species richness for each treatment in each replicate was determined as the total number of species caught on a transect pair during the trapping period each season. Species richness between treatments was then compared using the mean number of species for each treatment from the five replicates, using the same approach as for abundance analyses.

8.2.3 Biomass

Biomass for each treatment in each replicate block was calculated as the mass of all animals of each species caught on a transect pair, and all animals combined for each transect pair. Animals that escaped before weighing and/or sexing were given the average weight of that species on the transect pair. Generalized linear models were used for analyses, with a Gaussian distribution and a log-link, using biomass as the continuous response variable, year as a continuous predictor and treatment as a factor. We aimed to find the minimum adequate model though step-wise simplification and testing of the significance of interactions and the main effects, as for abundance.

8.2.4 Over-winter survival

For voles and Wood mice, we calculated over-winter survival as the percentage of those animals that were PIT-tagged in the Autumn that were re-captured the following Spring. Treatments were then compared where appropriate.

8.2.5 Habitat variables

Habitat variables were available for each transect pair and were tested for their ability to improve the fit of the models describing mammal abundance, biomass and species richness. Hedgerow berry counts (total fruits per 5 m^2 sampling area) were available for the Autumn

2006-10, but not the baseline Autumn of 2005. As there was no significant difference in berry densities between treatments and years (non-significant result not shown), however, we took the 5-year mean for each transect pair as the baseline data. For the hedgerow within an extent of 50 m surrounding each transect in 2007, we delineated the length, volume and area, mean height and its standard deviation, the mean width, and the volume of mature trees. Hedgerow and berry variables were added to the models only once the minimum adequate model had been achieved using the predictors of year and treatment, and if the residuals of this model had a significant correlation with any variable.

8.3 **Results**

Due to interference at the traps by Carrion Crows (*Corvis corone*) and Magpies (*Pica pica*), which had learnt to systematically pull out the traps to gain access to the food, trapping had to be abandoned on the CC and ELS transects in a single block in Spring 2011 (Year 5), resulting in a lower number of replicates for these treatments in this season. For all other treatments in the Spring and Autumn seasons, five replicate data points were derived for each sampling year, totalling 15 overall.

8.3.1 Abundance

8.3.1.1 Total abundance in Autumn

For Autumn periods, total counts of all mammal species increased on all treatments immediately after installation of the treatment margins in Year 0. The minimum adequate model (Figure 8.1) indicated no significant effect of treatment and no interaction between treatment and year. The model depicted a significant 102% annual increase of mean abundance on all treatments combined between Years 0 and 2 ($F_{57,58} = 34.39$, $P < 0.001$), but the fitted values indicated that abundance had stabilised between Years 3 and 5, with no significant further change by the end of the experiment. There was no relationship between abundance and the habitat variables tested.

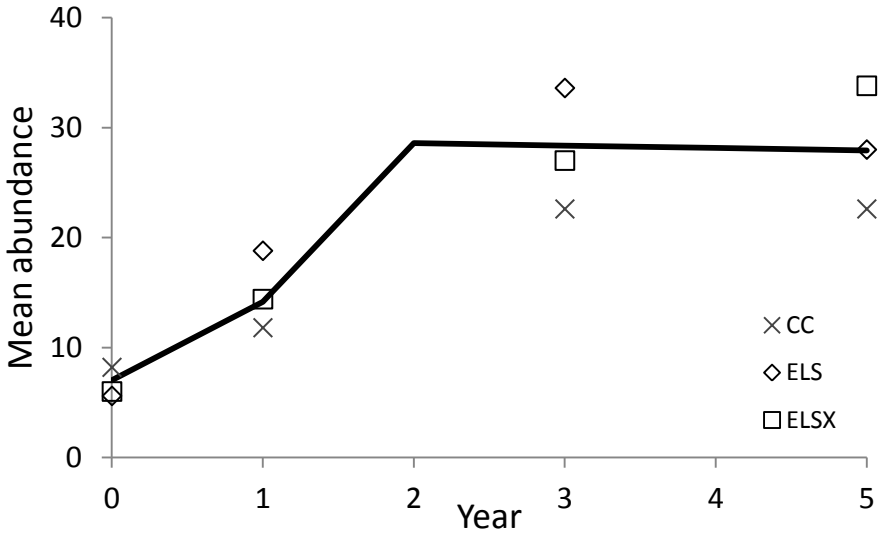


Figure 8.1 Fitted model of total abundance in Autumn.

8.3.1.2 Wood Mouse abundance in Autumn

The minimum adequate model for Wood Mouse in Autumn indicated no significant difference between treatments and no relationship with habitat variables (Figure 8.2). The model for combined treatments showed a statistically significant 66% annual increase in mean abundance between Years 0 and 2 ($F_{57,58}$ = 16.46, P < 0.001). Abundance then showed a non-significant annual 10% decline thereafter. By the end of the experiment in Year 5, however, Autumn mean abundance of Wood Mice had nevertheless increased by 102% compared to the baseline Year 0.

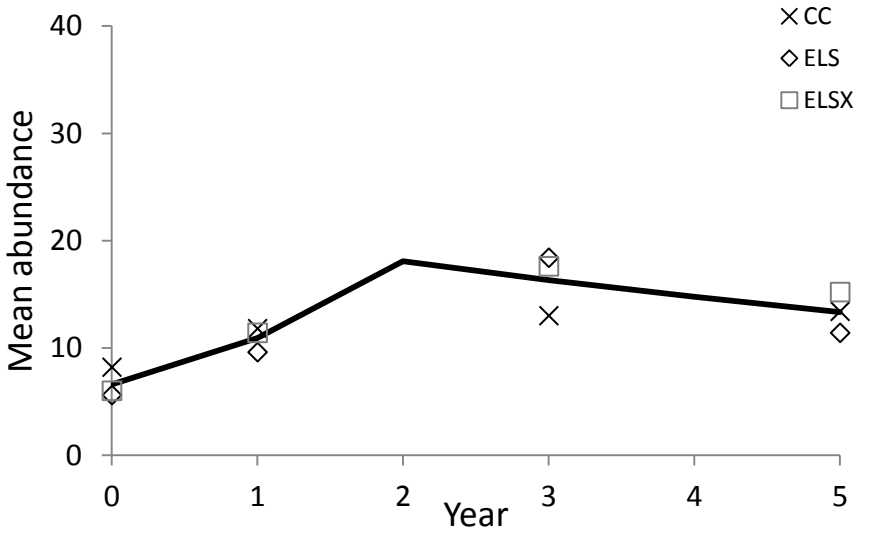


Figure 8.2. Fitted model of Wood Mouse abundance in Autumn.

8.3.1.3 Bank Vole abundance in Autumn

The minimum adequate model for Bank Vole in Autumn showed no significant interaction of treatment and time, but a significant treatment effect of 76% greater mean abundance on ELS and ELSX treatments compared to the CC control ($F_{57,58}$ = 7.47, P = 0.008) (Figure 8.3). Bank Voles were rare or absent on all treatments during the baseline Autumn (Year 0), but showed a significant 648% annual increase after treatment installation, between Years 0 and 2 ($F_{56,57}$ = 60.39, P < 0.001). The model suggested that mean abundance stabilised on all treatments in Year 2, with no significant change thereafter.

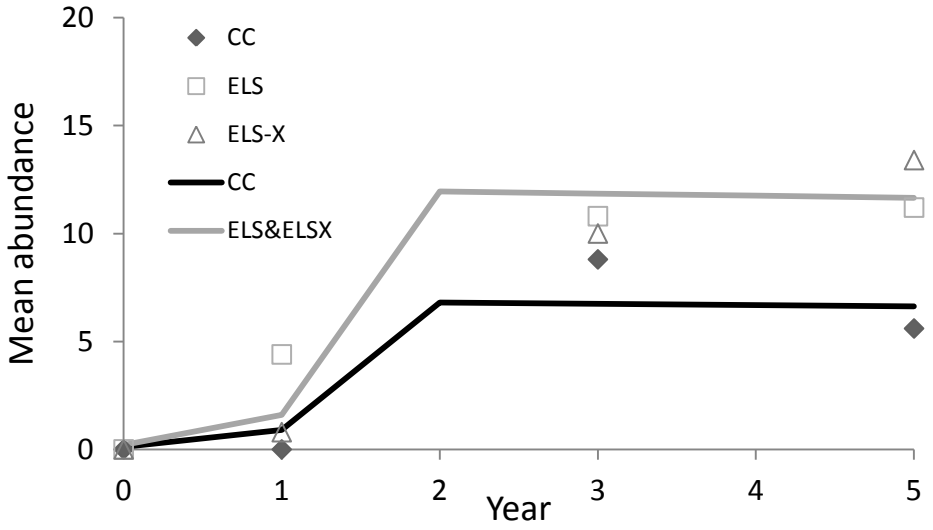


Figure 8.3. Fitted model of Bank Vole abundance in Autumn.

8.3.1.4 Field Vole abundance in Autumn

The minimum adequate model for Field Voles in Autumn depicted a statistically significant treatment effect of ELS and ELSX supporting 275% greater mean abundance than the CC control ($\chi^2_{57,58}$ = 12.73, P < 0.001), with a significant annual increase of 60% from a very low baseline on all treatments ($\chi^2_{57,58}$ = 36.51, P < 0.001) (Figure 8.4). Overall numbers were relatively low compared to Bank Vole and Wood Mouse.

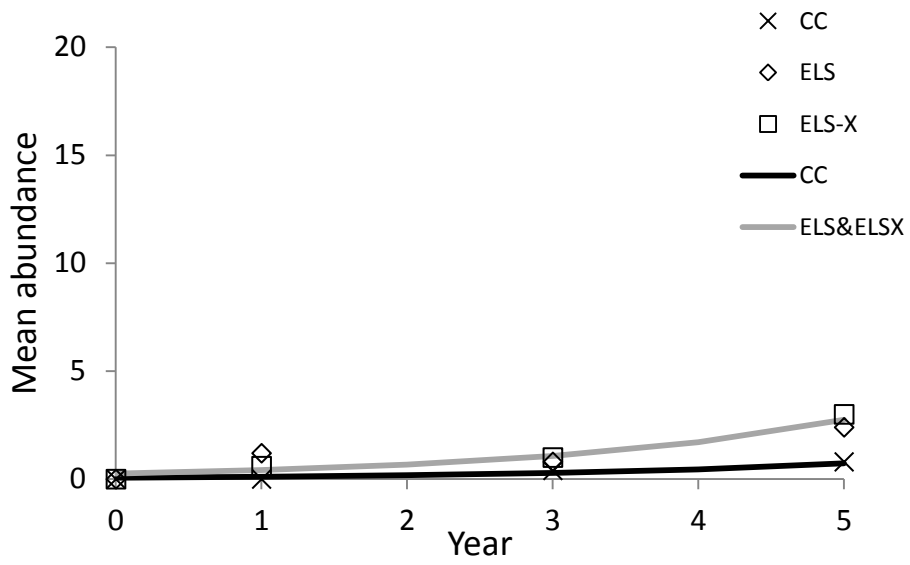


Figure 8.4. Fitted model of Field Vole abundance in Autumn.

8.3.1.5 Common Shrew abundance in Autumn

The minimum adequate model for Common Shrews in Autumn indicated a significant interaction of treatment and year ($\chi^2_{55,56}$ = 4.77, *P* = 0.029), with a greater mean abundance on ELS than either ELSX or CC, which did not differ (Figure 8.5). The difference between treatments varied from 1135% in Year 0 before converging towards an 81% difference in Year 5. There was a significant positive effect of berry count on all treatments ($\chi^2_{55,56}$ = 16.09, *P* < 0.001), with every 100 berries per 5 m^2 of hedgerow inflating Common Shrew abundance by an estimated mean 0.18 animals each year. As Common Shrews are insectivorous, however, this result may have had an indirect cause.

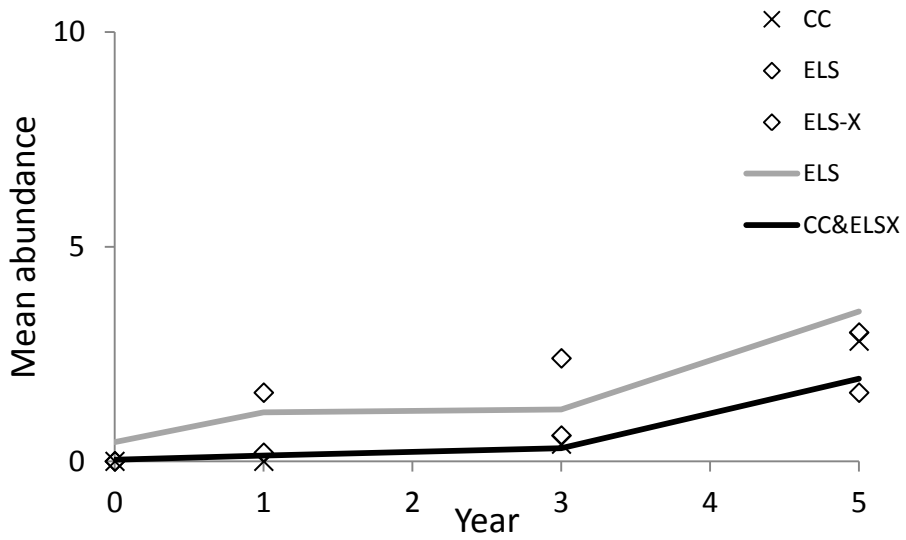


Figure 8.5. Fitted model of Common Shrew abundance in Autumn.

8.3.2 Total abundance in Spring

For total mammal mean abundance in Spring, there was a significant interaction of treatment and year ($F_{53,54}$ = 21.12, P < 0.001), with ELS and ELSX being distinct from CC but not each other (Figure 8.6). There was also a significant small positive effect of the previous Autumn berry count across all treatments ($F_{53,54}$ = 35.93, P = 0.018), with every 100 berries adding 0.1 animals to mean abundance. The minimum adequate model indicated that abundance was significantly greater on the CC treatment compared to the combined ELS and ELSX treatments in Years 0 to 3, peaking in Year 1 but showing an overall decline of 22% between Year 0 and year 5. Estimated mean abundance on ELS and ELSX treatments showed an increase of 656% between Years 0 and 5, being substantially greater on these treatments than on CC by the final Spring.

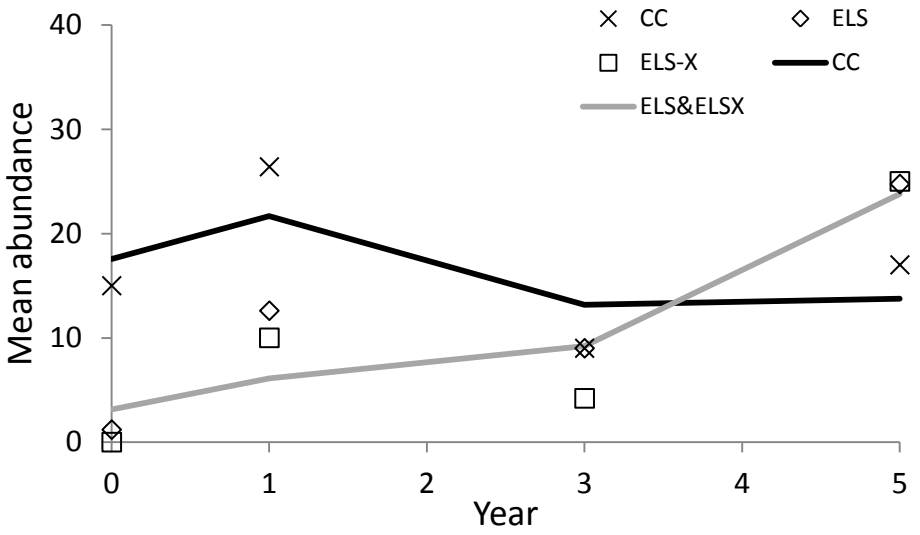


Figure 8.6. Fitted model of total abundance in Spring.

8.3.2.1 Wood Mouse abundance in Spring

The minimum adequate model for Wood Mouse in Spring showed a significant difference between the CC control and the ELS and ELSX treatments ($F_{55,56}$ = 57.30, P < 0.001) (Figure 8.7). Abundance on ELS and ELSX treatments was lower than that on CC, and all showed a statistically significant 25% annual decrease ($F_{55,56}$ = 13.78, P < 0.001). Berry count during the previous Autumn had a near significant positive effect on Wood Mouse mean abundance on all treatments ($F_{54,55}$ = 15.59, P = 0.058), but this was not retained in the final model.

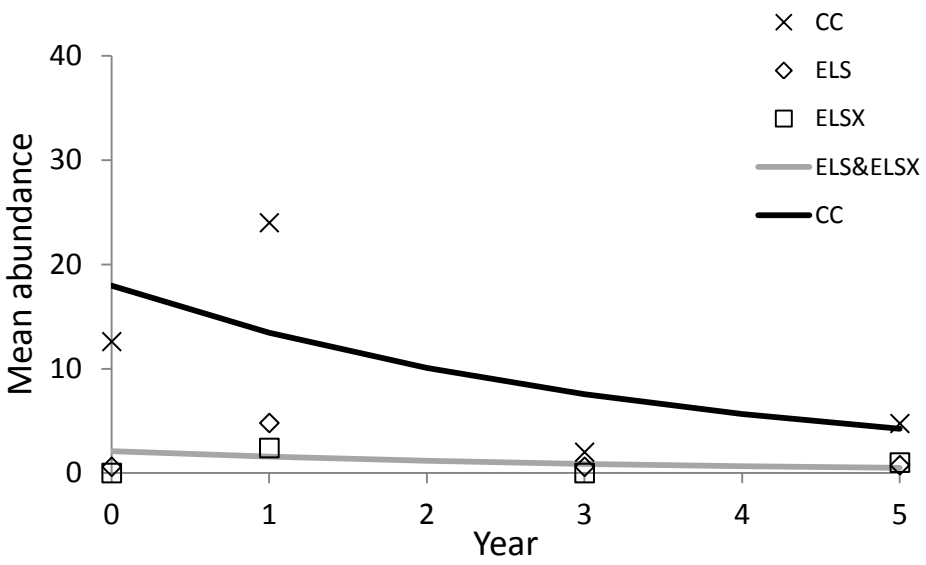


Figure 8.7. Fitted model of Wood Mouse abundance in Spring.

8.3.2.2 Bank Vole abundance in Spring

There was a significant interaction between treatment and year for the modelled Bank Vole mean abundance in Spring ($F_{54,55}$ = 26.02, P = 0.008) (Figure 8.8). While mean abundance increased on all treatments over time, the model estimated that CC and ELS outperformed ELSX by 847% in Year 0, before gradually converging by Year 4, with abundance on ELSX eventually finishing 63% higher than other treatments by Year 5.

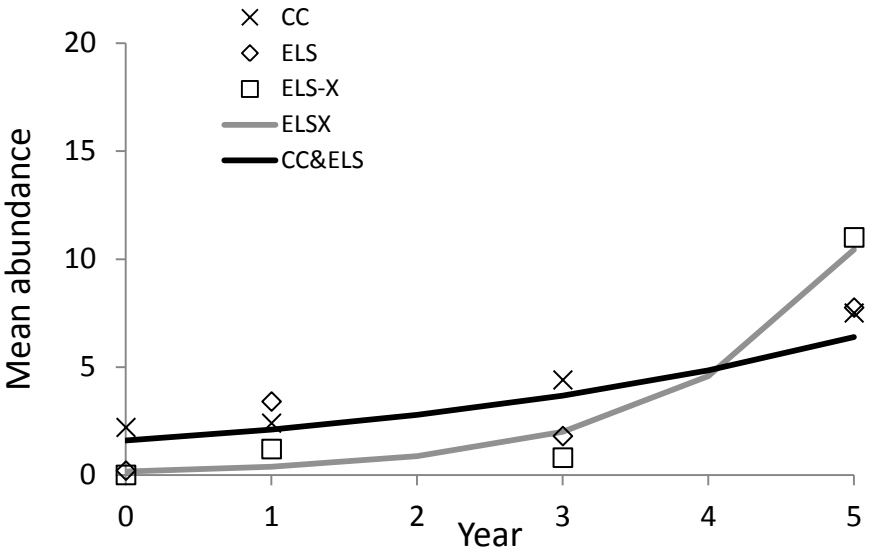


Figure 8.8. Fitted model of Bank Vole abundance in Spring.

8.3.2.3 Field Vole abundance in Spring

The minimum adequate model for Field Vole abundance in Spring found a significant treatment effect (Figure 8.9), with mean abundance on ELS and ELSX being 300% higher

than on CC ($F_{54,55}$ = 15.16, P < 0.001). There was a significant increase on all treatments over time ($F_{54,55}$ = 88.84, P < 0.001), and a significant positive effect of the berry count from the previous Autumn ($F_{54,55}$ = 36.19, P < 0.001), with an estimated 0.2 animals being added to mean abundance for every 100 berries per 5 m^2 of hedgerow.

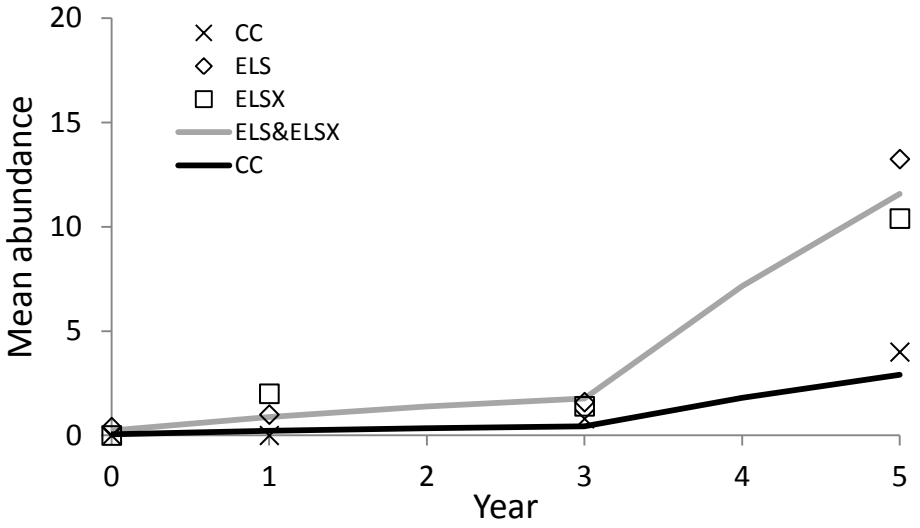


Figure 8.9. Fitted model of Field Vole abundance in Spring.

8.3.2.4 Common Shrew abundance in Spring

Modelling the Spring abundance of Common Shrews revealed no significant effect or interactions of treatment, year or habitat variables, although the minimum adequate model indicated a non-significant 19% annual increase over time, with ELS and ELSX performing 279% better than CC, but not each other (Figure 8.10).

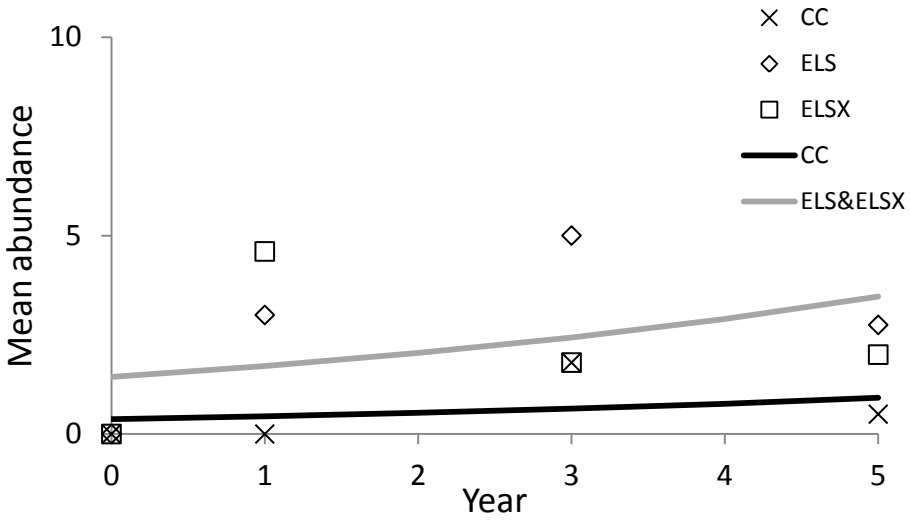


Figure 8.10 Fitted model of Common Shrew abundance in Spring.

8.3.3 Abundance summary

- In Autumn, changes in abundance of the total mammal population, largely driven by the common mammal species (Wood Mouse and Bank Vole), predominantly occurred within the initial two years after treatment installation, but were maintained at this level for a further 3 years until the end of the experiment. The scarcer species, Field Vole and Common Shrew, showed a continuous increase during the experiment.
- There was no treatment effect for overall mammal abundance in Autumn, or for Wood Mouse, but Bank Vole and Field Vole were all more abundant on ELS and ELSX than CC (ELS only for Common Shrew). ELSX did not perform better than ELS for any species in Autumn.
- In Spring, total mammal abundance did not increase on the CC control, but increased on ELS and ELSX treatments, with a further positive effect of hedgerow berry yield in the previous Autumn (likely increasing the over-winter food supply). Wood Mouse Spring abundance declined over time, while the abundance of Bank Voles and Field Voles increased, but there was no significant trend for Common Shrews.
- Treatment effects were more species-specific in Spring compared to Autumn, although by the end of the experiment Bank Vole showed a greater overall abundance on ELSX compared to other treatments, and Field Voles were more abundant on ELS and ELSX than on CC throughout the experiment, with berry yield also having a positive effect on this species.

8.3.4 Species richness

Overall, 10 species of small mammal were caught during the experiment: Wood Mouse, Bank Vole, Field Vole, Harvest Mouse, House Mouse, Common Shrew, Water Shrew, Pigmy Shrew, Brown Rat (*Rattus norvegicus*) and Common Weasel (*Mustela nivalis*). The latter two (caught once) are excluded from species richness totals.

8.3.4.1 Species richness in Autumn

Wood Mouse was the only species encountered on the estate at the beginning of the experiment (Autumn of Year 0), being present on all treatments. An increase in the mean number of species captured was observed on ELS and ELSX margins during the Autumn of

Year 1, but not on CC margins until the next sampling period in Year 3, with little change to Year 5 (Figure 8.11).

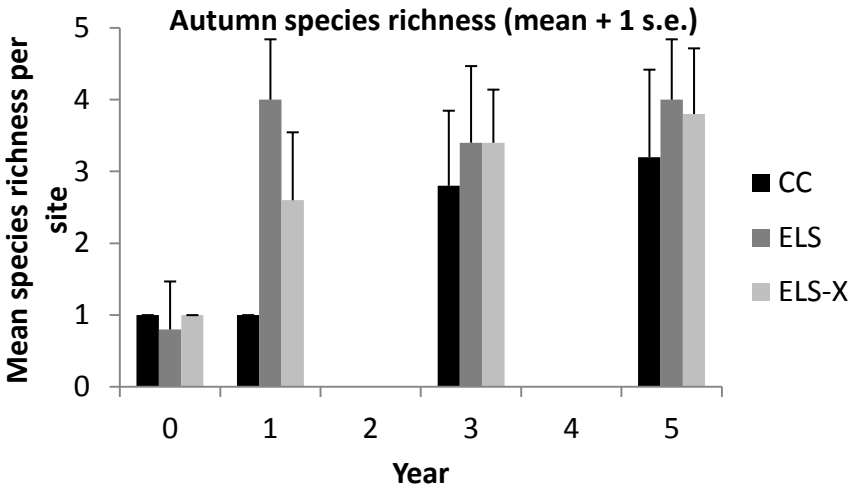


Figure 8.11. Variation in Autumn species richness on treatments over time.

The minimum adequate model for Autumn species richness (Figure 8.12) identified a significant treatment effect, with richness being 44% higher on ELS and ELSX than the CC control ($\chi^2_{56,57}$ = 4.13, P = 0.042). There was also a significant effect of Year, although this was discontinuous; after a significant increase in mean species richness on all treatments between the baseline Year 0 and Year 1 ($\chi^2_{56,57}$ = 13.26, P < 0.001), the subsequent increases indicated by the model for Years 1-5 were not statistically significant.

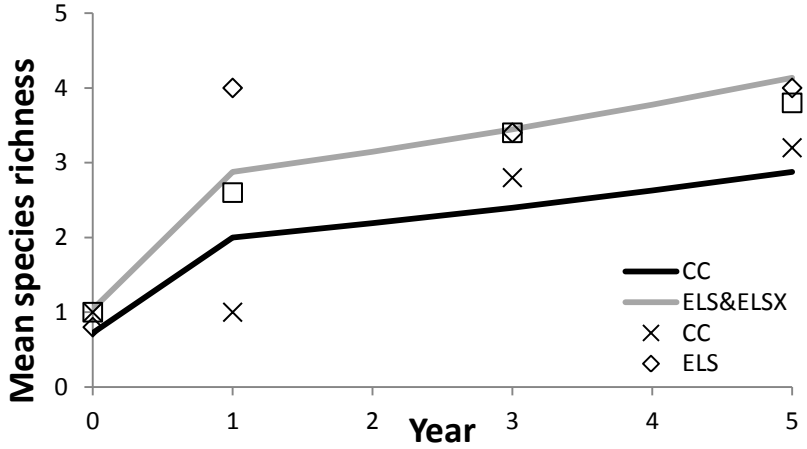


Figure 8.12 Minimum adequate model for Autumn species richness on treatments over time

8.3.4.2 Species richness in Spring

Spring species richness was variable on and between treatments over time (Figure 8.13). Wood Mouse, Bank Vole and Harvest Mouse were present in the study area in Year 0, increasing to a total of 6 species by the Spring of Year 2 (with the addition of Field Vole and Common, Water and Pigmy Shrew, but absence of Harvest Mouse), and 7 in Year 5 (as Year 3, plus House Mouse).

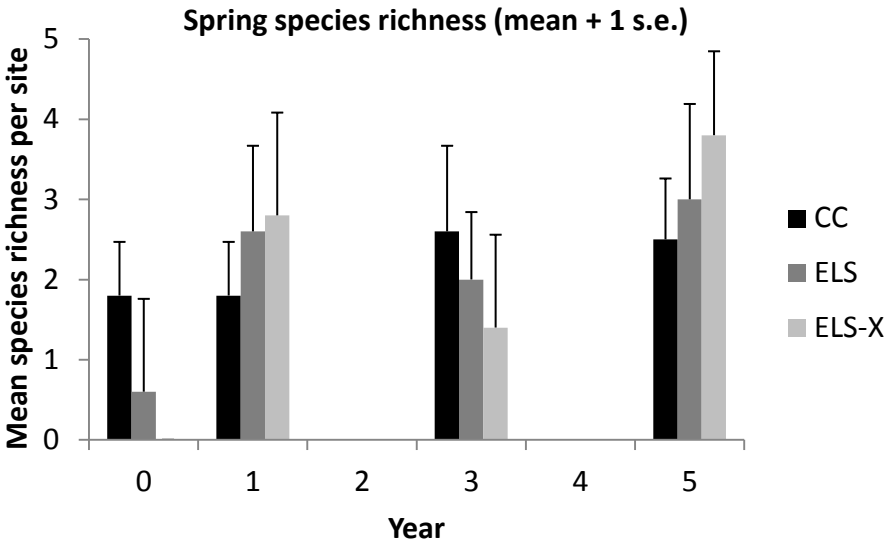


Figure 8.13 Variation in Spring species richness on treatments over time.

The minimum adequate model for species richness in Spring showed no treatment effect (Figure 8.14), although there was a significant 24% annual increase over time ($\chi^2_{55,56} = 4.46$, $P = 0.035$) which was moderated by a significant effect of the berry count from the previous Autumn ($\chi^2_{55,56} = 17.00$, $P < 0.001$), with an additional 0.1 species per 100 berries in 5 m^2 of hedgerow on all treatments. The relatively low berry yield in Year 3 was indicated by the low (8%) rate of increase in mean species richness between Years 1 and 3, with larger yields in Years 1 and 5 being associated with a more obvious increase in the fitted values of the model.

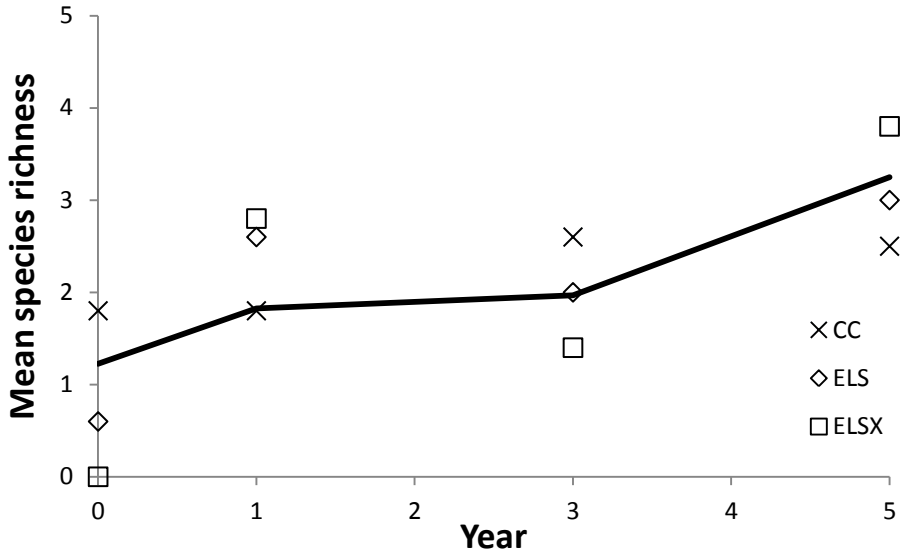


Figure 8.14 Minimum adequate model for Spring species richness on treatments over time.

8.3.4.3 Species richness summary

- Mean species richness increased on all treatments in both seasons after treatment establishment, from a very low baseline. In Autumn, this increase was primarily achieved within the first year after treatment installation, while in Spring the increase was more gradual and continuous over time.
- Species richness was 44% higher on ELS and ELSX compared to CC in Autumn, but there was no treatment effect in Spring.
- Autumn berry yield had a significant positive effect on mean species richness the following Spring, but did not influence richness in Autumn.

8.3.5 Biomass

8.3.5.1 Total biomass in Autumn

The minimum adequate model describing the total biomass of all animals in Autumn (Figure 8.15) identified a significant treatment effect of the ELS and ELSX treatments holding 33% more biomass than CC throughout the duration of the experiment ($F_{56,57}$ = 5.41, P = 0.024). Modelled biomass showed a highly significant 105% annual increase on all treatments between Years 0 and 2 ($F_{56,57}$ = 31.20, P <0.001), but with no significant change thereafter ($F_{56,57}$ = 0.10, P = 0.756).

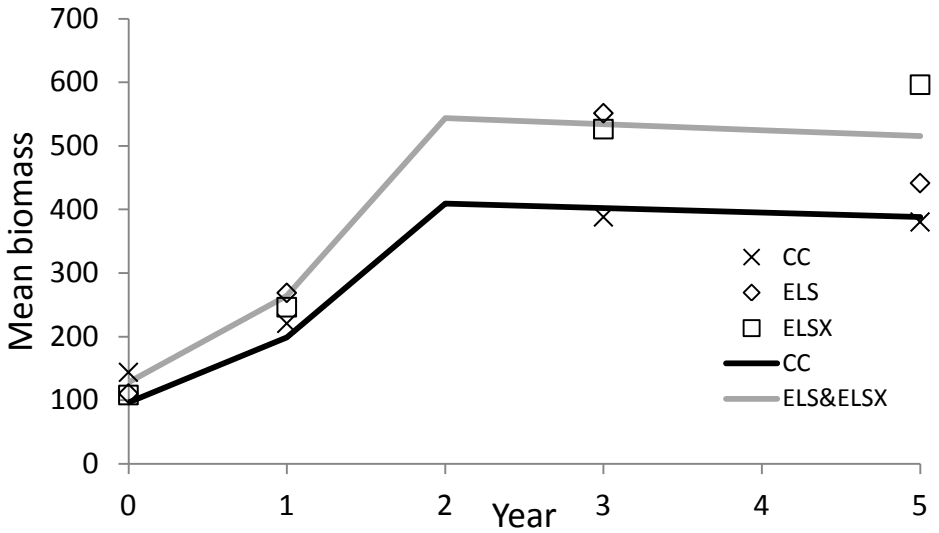


Figure 8.15 Modelled mean total biomass of all animals in Autumn.

8.3.5.2 Wood Mouse biomass in Autumn

No significant treatment effects were evident in the minimum adequate model for Wood Mouse biomass in Autumn (Figure 8.16), but a highly significant annual increase of 63% was modelled on all treatments between Years 0 and 2 ($F_{57,58}$ = 13.56, P < 0.001), followed by a non-significant 8.5% annual decrease between Years 2 and 5 ($F_{57,58}$ = 1.51, P = 0.224). Despite the decline in the later years, the overall modelled increase in Wood Mouse biomass between Years 0 and 5 was 103%.

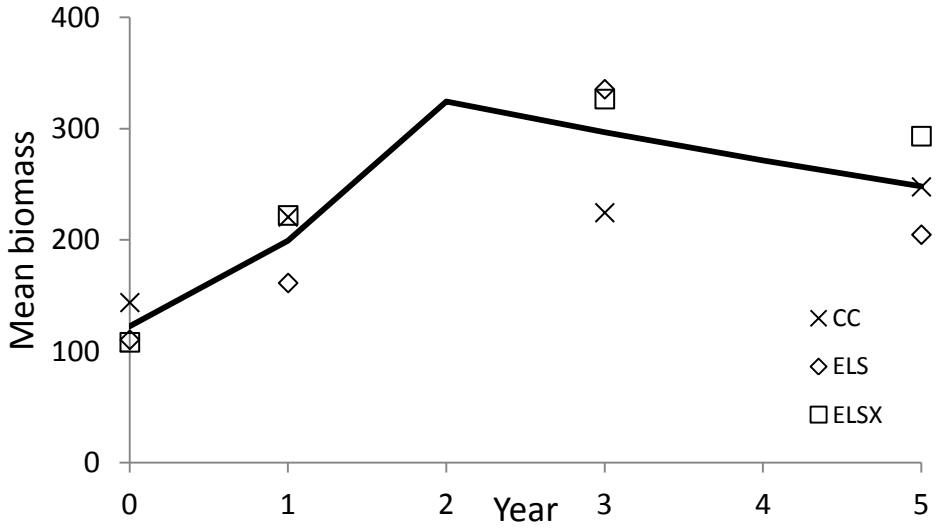


Figure 8.16 Modelled mean biomass of Wood Mice in Autumn.

8.3.5.3 Bank Vole biomass in Autumn

The model of Bank Vole Autumn biomass (Figure 8.17) mirrored that of abundance, with a significant treatment whereby mean abundance on ELS and ELSX was 57% greater than the CC control ($F_{56,57}$ = 5.39, P = 0.024). All treatments showed a significant 487% annual

increase between the very low baseline at Year 0 and Year 2 ($F_{56,57}$ = 28.75, P < 0.001), with no significant change thereafter.

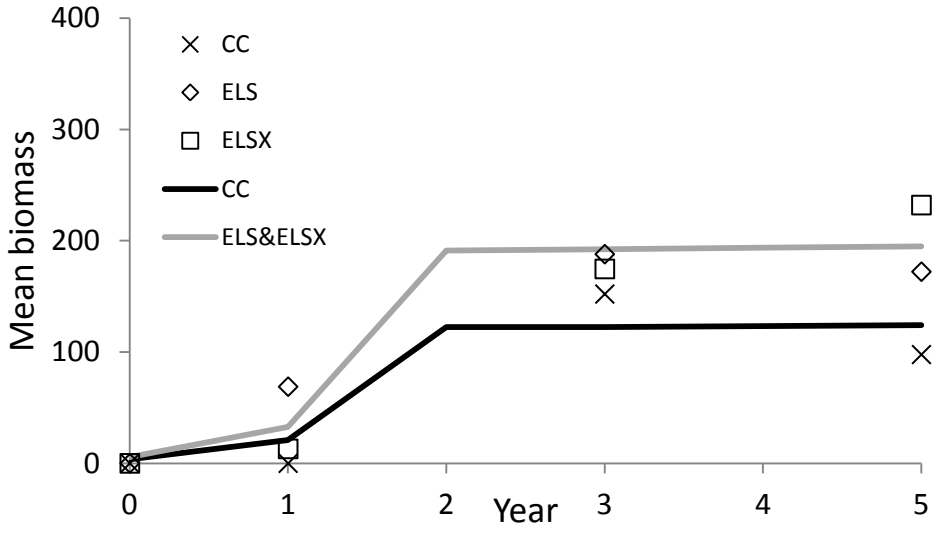


Figure 8.17 Modelled mean biomass of Bank Voles in Autumn.

8.3.5.4 Field Vole biomass in Autumn

The model of Field Vole Autumn biomass (Figure 8.18) showed a significant treatment effect with modelled mean abundance on ELS and ELSX being 215% greater than on the CC control ($F_{57,58}$ = 7.70, P = 0.008). There was also a significant effect of time, with an annual increase of 61% on all treatments ($F_{57,58}$ = 22.00, P < 0.001).

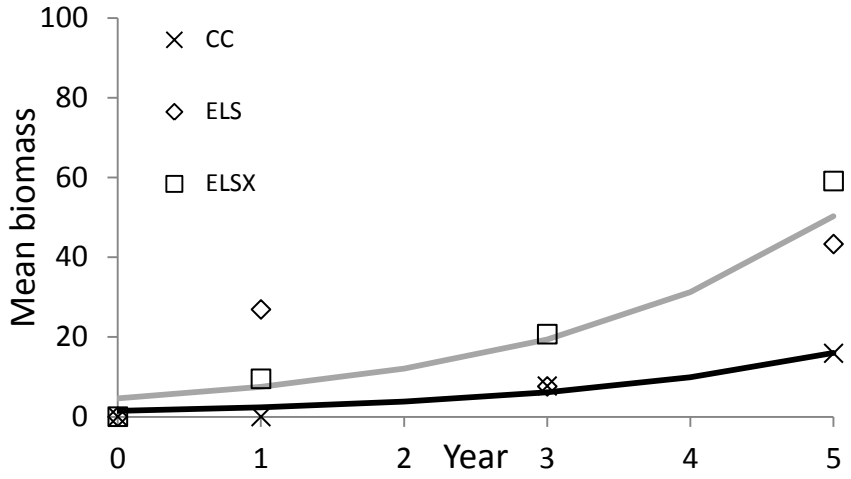


Figure 8.18. Modelled mean biomass of Field Voles in Autumn.

8.3.5.5 Common Shrew biomass in Autumn

The minimum adequate model for Common Shrew in Autumn (Figure 8.19) showed no treatment effects, but a significant annual increase of 48% on combined treatments ($F_{58,59}$ = 10.31, P = 0.002). By Year 5, Common Shrew biomass had increased on all treatments since the baseline Year 0, by a modelled mean of 602%.

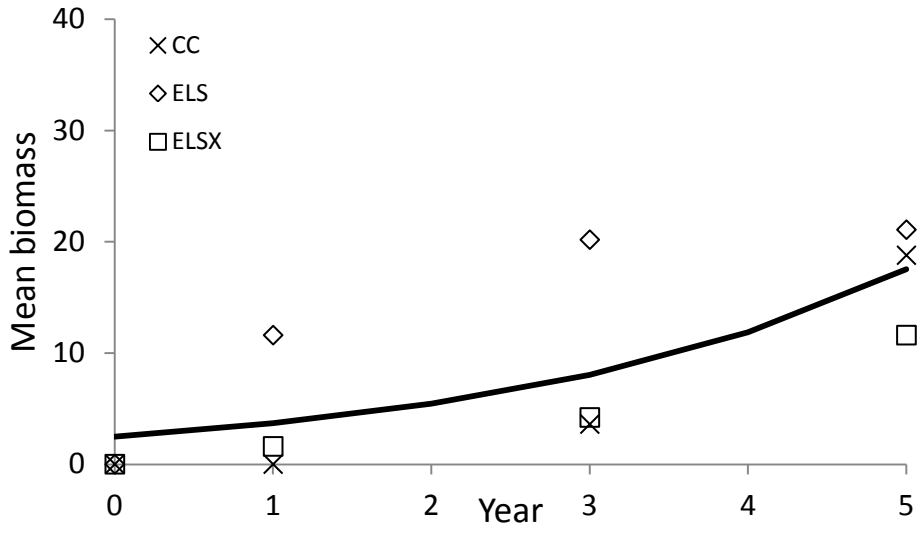


Figure 8.19 Modelled mean biomass of Common Shrews in Autumn.

8.3.5.6 Total biomass in Spring

The minimum adequate model for total biomass in Spring (Figure 8.20) depicted a significant interaction of time and treatment ($F_{54,55}$ = 26.49, P < 0.001), with ELS and ELSX treatments exhibiting a 70% annual increase from a very low baseline, while biomass on the CC control began at a comparatively high baseline but showed a non-significant 6% annual decline. By the final Year 5, the modelled mean biomass on ELS and ELSX treatments was 74% greater than on the CC control.

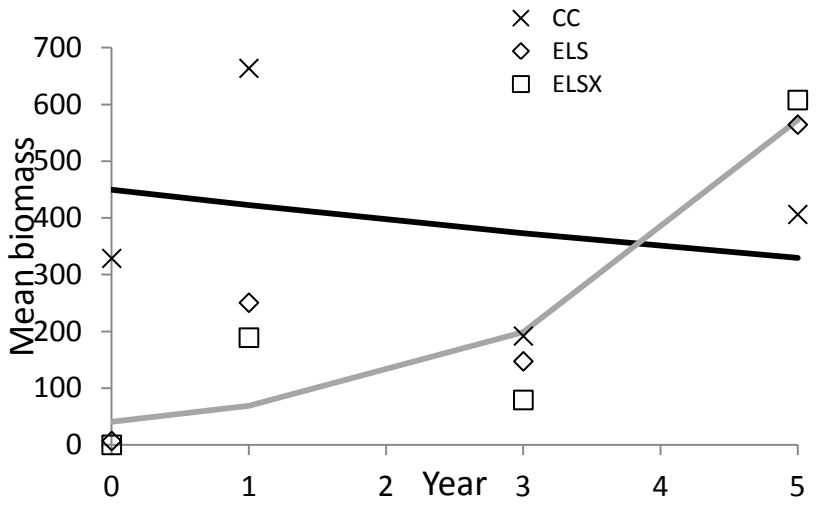


Figure 8.20 Modelled mean total biomass of all animals in Spring.

8.3.5.7 Wood Mouse biomass in Spring

The minimum adequate model for Wood Mouse biomass in Spring contrasted with that for Autumn, with a significant treatment effect ($F_{54,55}$ = 202.07, P < 0.001) indicating a mean abundance that was 762% greater on the CC control than the ELS and ELSX treatments, which did not differ from each other (Figure 8.21). There was a significant effect of time on mean abundance on all treatments, with an annual increase of 120% between Years 0 and 1 ($F_{54,55}$ = 48.09, P < 0.001), followed by an annual decline of 64% thereafter ($F_{54,55}$ = 148.34, P < 0.001).

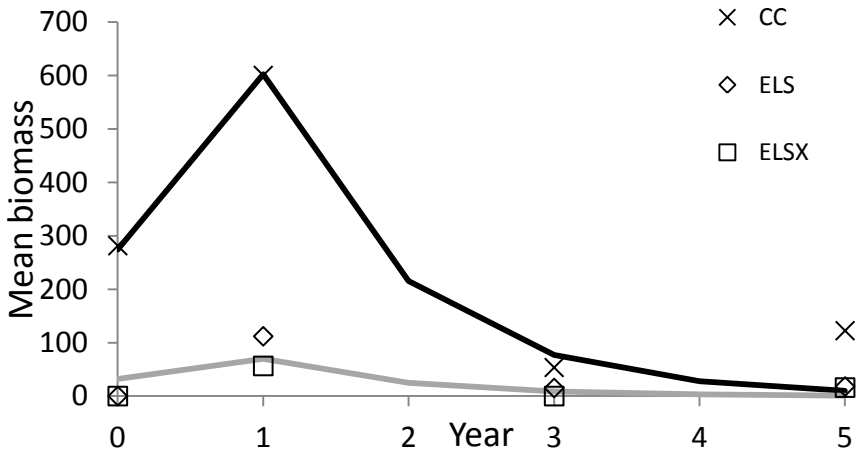


Figure 8.21 Modelled mean biomass of Wood Mice in Spring.

8.3.5.8 Bank Vole biomass in Spring

The model for Bank Vole biomass in Spring differed from that for abundance in detecting no significant effect of treatment, although there was a highly significant effect of time ($F_{56,57}$ = 34.08, P < 0.001) modelling an annual increase of 68% from a low baseline on all treatments (Figure 8.22).

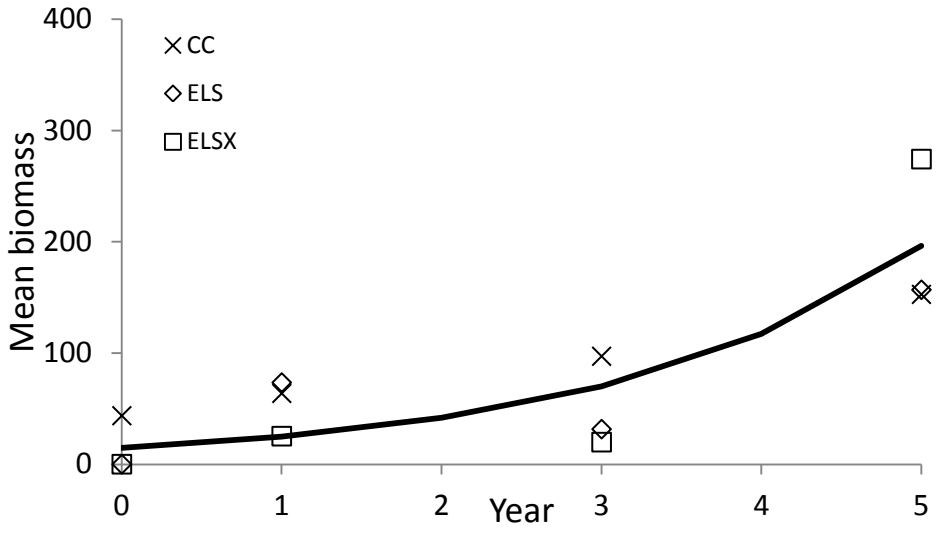


Figure 8.22 Modelled mean biomass of Bank Voles in Spring.

8.3.5.9 Field Vole biomass in Spring

Field Vole biomass in Spring largely mirrored that of abundance, with a significant effect of treatment isolating lower biomass on the CC control from higher values on both ELS and ELSX ($F_{55,56}$ = 7.51, P = 0.008), which were some 152% higher. There was a significant effect of time acting upon all treatments ($F_{55,56}$ = 45.75, P < 0.001), with a modelled 158% annual increase from a baseline approaching zero (**Error! Reference source not found.**). This attern may have been skewed, however, by a likely 'vole year' in Year 5, in which Field Vole numbers were at a widespread cyclical peak.

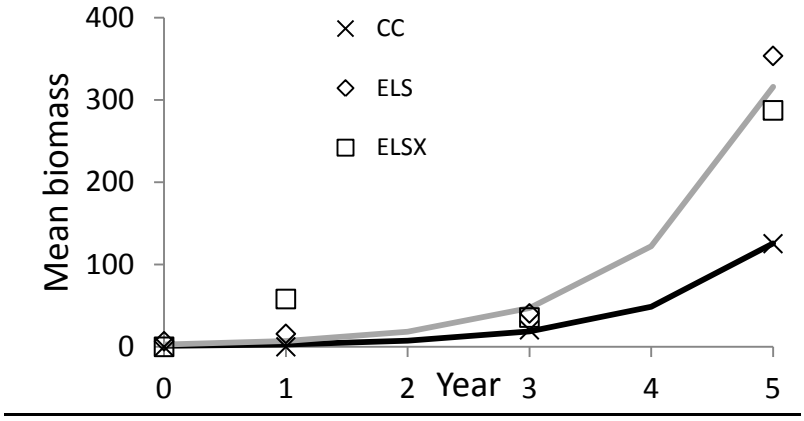


Figure 8.23 Modelled mean biomass of Field Voles in Spring.

8.3.5.10 Common Shrew biomass in Spring

The minimum adequate model for Common Shrew biomass in Spring contained no significant parameters, although a non-significant upward trend over time was present in the data (Figure 8.24).

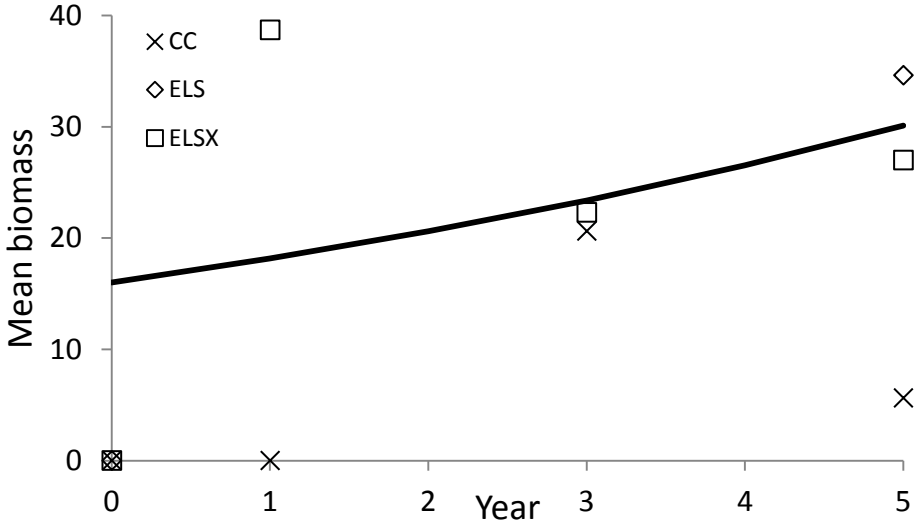


Figure 8.24. Modelled mean biomass of Common Shrew in Spring.

8.3.5.11 Biomass summary

- Trends in biomass largely followed those of abundance, but with notable differences in treatment effects.
- Like total mammal abundance, Autumn total biomass showed a significant increase (105% per annum) in the initial two years after treatment installation, before settling at a higher level than the baseline, which was maintained til the end of the experiment.

- This pattern was replicated for the two common mammal species in Autumn, Bank Vole and Wood Mouse. Common Shrew and Field Vole showed a continuous increase in mean biomass in Autumn throughout the experiment.
- Treatment effects were evident in Autumn, with biomass being greater on ELS and ELSX for total mammals (by 33%), Bank Vole and Field Vole. There was no treatment effect for Wood Mouse or Common Shrew.
- Total mammal biomass in Spring increased by 70% per annum on ELS and ELSX treatments, but not on the CC control. Biomass on CC began from a higher baseline, due to a relatively large presence of Wood Mice, but showed little change.
- Wood Mouse biomass showed a complex pattern in Spring, with an increase during the first year after treatment installation before a subsequent and continuous decline, though biomass was greater on CC than ELS and ELSX throughout. Bank Vole biomass showed no treatment effect but a continuous increase throughout the experiment, while Field Vole also showed a continuous increase, with a significant effect of ELS and ELSX holding greater biomass than CC. Common Shrew showed no significant trends.
- Hedgerow habitat and berry yield had no effect on biomass.

8.3.6 Over-winter survival

The number of animals PIT-tagged in Autumn and recaptured in the subsequent Spring was very low (an average of just 10 per year, or 3% of the Autumn catch). Numbers were too small for a meaningful analysis of a treatment effect, e.g. only 16, 9 and 14 retraps on CC, ELS and ELSX respectively during the whole study, spread over three species. As such, no inferences could be drawn.

8.4 **Overall summary**

- There was a substantial and rapid farm-scale effect of an increase in mammal abundance, biomass and species diversity over time, after ELS and ELSX margins were established.
- In Autumn, effects predominantly occurred within the initial two years after treatment installation, but were maintained at this level throughout the rest of the experiment.

- There was no treatment effect on total abundance of mammals in Autumn, but the species richness and biomass of these animals was greater on ELS and ELSX than on CC, although the two treatments did not differ significantly from each other.
- In Spring, increases in total mammal abundance and biomass were more gradual and continuous than in Autumn, with no levelling off of the effects observed by the end of the experiment.
- Biomass and abundance was initially higher on CC than other treatments in Spring, indicating a slow establishment of the habitat or greater mortality on ELS and ELSX margins.
- The abundance and biomass of Wood Mice generally decreased between subsequent Spring periods, but species richness increased on all treatments and Wood Mice were largely replaced by an increase in the abundance and biomass of Field Voles and Bank Voles.
- Berry yield in the previous Autumn had a positive effect on species richness and total abundance in Spring.
- Over-winter survival appeared to be very low for individual voles and Wood Mice on all treatments.
- The experiment indicated that ELSX did not perform significantly better than the ELS treatment, although there was strong evidence that these treatments performed better than the CC control in increasing mammal abundance, species richness and biomass in Spring and Autumn.
- Significant increases in mammal abundance, species richness and biomass on the CC control also indicated that a whole farm or landscape-scale effect was achieved without having to place ELS or ELSX treatment margins on all fields. The density of ELS and ELSX margins adopted in the experiment appeared capable of enhancing the mammal communities of neighbouring conventional margins.

9 Birds

In the UK, agricultural intensification since the 1940s has caused farmland bird populations to undergo large-scale, well documented decline and range contractions. As a result farmland birds are one of the top priorities for biodiversity conservation in the UK and have

been adopted by the Government as one of a suite of sustainability indicators. Measures of success are based upon the Farmland Bird Index (FBI), a composite indicator of 19 national breeding farmland bird species used to monitor the effectiveness of measures to improve the sustainability of UK agriculture (Gregory *et al.* 2004). The FBI was adopted in the late 1990s and, until recently, has continued to decline throughout this period. The species mainly responsible being a rather disparate group comprising five resident species (Stock Dove *Columba oenas*, Greenfinch *Carduelis chloris*, Linnet *Carduelis cannabina*, Lapwing *Vanellus vanellus*, Kestrel *Falco tinnunculus*) and two summer migrants (Whitethroat *Sylvia communis* and Yellow Wagtail *Motacilla flava*) (JNCC 2009). Various reasons related to key AES options have been suggested for this downturn, but given the level of uncertainty, there is a clear need to investigate the effectiveness of ES options as they are applied on farmland (Davy *et al.* 2009).

From the results of previous studies (e.g. Boatman *et al.* 2000, Henderson *et al.* 2004, Stoate *et al.* 2004), we hypothesized that food patches would attract greater numbers of birds than cropland, thus increasing bird abundance in winter and having the potential to enhance the size of the breeding population. We examined responses of both individual species and groups of species and how spatial scale and variation in the data affected our ability to detect differences between treatments compared to changes at a whole farm-scale. We also investigated the effects of patch seed crop size on bird usage, with an expectation of a positive relationship, and evaluated the use made by birds of other types of prescribed margin and patch habitat types in late winter.

9.1 **Objectives**

In this Section we describe the bird data collected over the five years of the project from the winter of 2005/06 to that of 2010/11. Data were collected as follows:

a) Breeding bird abundance and distribution in the four years of 2006, 2007, 2009 and 2010.

b) Breeding bird productivity in the two years of 2008 and 2010.

c) Winter bird abundance and distribution in the years of 2005/06, 2006/07, 2009/10 and 2010/11.

d) Bird use of dedicated winter bird food patches (and controls) in the winter of 2007/08. (Data for the patches alone were also collected in the winters of 2008/09 and 2009/10 but are not included in the analysis here – see text below)

e) Bird use of other prescribed habitats in late winter/early spring in 2008.

f) Great Tit *Parus major* and Blue Tit *Cyanistes caeruleus* breeding success in the four years 2007, 2008, 2009 and 2010.

Results for d) and e) have been published (Hinsley *et al.* 2010a and 2010b); copies of the content of the papers are included as Appendices 12.4 and 12.5 and are referred to as required.

The baseline year for the winter bird data was 2005/06 and for the breeding bird and patch data, it was spring 2006. For the tit breeding data, boxes were first provided in the late winter/early spring of 2007 (funded by Syngenta).

9.2 **Methods**

9.2.1 Breeding Birds

Breeding birds were recorded at the scale of each Treatment (i.e. Cross Compliance, CC; Entry-Level Scheme, ELS; Entry-Level Scheme Extra, ELS-X) in each of the five replicate Blocks. Birds were monitored using hedgerow transects located in the interior of each Treatment (with one exception where *c.* one third of the transect in one CC Treatment comprised a Treatment boundary; Appendix 12.1). Within each Treatment, the hedges were usually contiguous with a few instances of interspersed small copses, clumps of bushes/trees and gaps. Transect length ranged from 636 m to 1,803 m, the mean being 1077 m and the total length of hedgerow censused 16.15 km. The lengths of all 15 transects (3 Treatments x 5 Blocks), and selected structural details derived from the LiDAR data, are given in Table 9.1. All birds in the hedges, hedgerow trees and within 10 m of either side were recorded by spot mapping (Bibby *et al.* 2005), bird locations and activities being recorded on large-scale maps which included the locations of all hedgerow trees and other landmarks. Particular attention was paid to behaviour indicative of breeding. Four visits (April, May, June and early July) were made in each breeding season. Visits were made during the morning, starting shortly after dawn and finishing before midday, with the order in which individual transects were recorded being reversed on alternate visits. Visits were usually completed in three or four days depending on the weather. Weather likely to suppress activity and detection, e.g. strong winds and rain, were avoided. All bird records were later digitised using the LiDAR canopy height model within a geographic information

system (ArcMAP v9.3 © 2008 ESRI Inc.). Territory locations were estimated using observer judgement based on recorded bird behaviour to assign each record to a territory (Bibby *et al.* 2005), and then generating minimum convex polygons to represent approximate territory boundaries. A 10 m buffer was placed around the resultant polygons, in order to account for possible error when census mapping and digitizing locations. Territories estimated to occur wholly or substantially (more than two thirds minimum convex polygon area) within each Treatment were counted for each species. The total numbers of territories per kilometre per transect were then calculated for each individual species and also for the three groups of 'all species', 'all resident species' and 'granivorous species', and for the latter two groups without the most numerous species, Chaffinch *Fringilla coelebs*. Granivorous species comprised Chaffinch, Linnet *Carduelis cannabina*, Yellowhammer *Emberiza citronella* and Reed Bunting *Emberiza schoeniclus*.

9.2.2 Breeding Bird Productivity

Breeding birds were recorded as above but each transect was recorded on a weekly basis for ten weeks starting at the beginning of June and ending at the beginning of August in each of the two seasons. This period was selected to coincide with the peak of the finch and bunting breeding season. Although breeding usually extended well into August, the census was timed to finish just as the harvest began to avoid differential effects on birds breeding in different transects depending on when the crops in adjacent fields were harvested. During each of the ten visits, special attention was paid to any breeding activity and especially to the presence of newly fledged young and dependent juveniles. Although the finches and buntings were of special interest, all breeding bird species were recorded. However, Great Tits and Blue Tits were excluded from the analysis because data for these two species were collected in detail separately (see below) and because the time period used did not coincide with their main breeding period. Territories were digitised as before. Fledged young and dependent juveniles were used as a measure of productivity within each transect and were combined non-additively at the level of individual territories, i.e. for a given territory, dependent juveniles recorded shortly after the presence of fledged young were deemed to be the same individuals and thus productivity was recorded as one brood. If dependent juveniles were detected in the absence of such fledged young, or only fledged young were detected, each of these two instances would again be recorded as one brood.

Overall productivity for each individual species and for the three groups of 'all species', 'all resident species' and 'granivorous species' was then expressed as the number of broods per kilometre and the number of broods per territory for each transect. It should be noted that these assessments of productivity included three additional species, Greenfinch *Carduelis chloris*, Bullfinch *Pyrrhula pyrrhula* and Goldfinch *Carduelis carduelis*, which were not included in the breeding bird (a) or winter bird (c) data. These three species were comparatively rare and were not adequately recorded by the four and three visit protocols of the breeding season and winter censuses respectively. They were better represented in the ten visit productivity data and thus are included here. It should also be noted that territory estimation based on ten visits rather than four is responsible for the greater numbers of territories delimited within the productivity dataset for 2010 compared to that for the four-visit 2010 breeding season (see Results). This difference in methodology means that the productivity and breeding season datasets should not be compared directly, or combined, without due caution.

9.2.3 Winter Birds

The birds present in winter were also recorded at the scale of each treatment (i.e. CC, ELS and ELS-X) using the same 15 hedgerow transects (Table 9.1) used for the breeding birds. All birds, and their activities, in the hedges, hedgerow trees and within 10 m of either side were again recorded by spot mapping on large-scale maps. Birds in the transects were recorded on three visits (November, December and January) in each of the winters. Counts began a little after dawn and finished at least two hours before dusk and were not undertaken in weather likely to depress bird activity. Each count was usually completed within two days, occasionally three in the event of poor weather. The bird records were again digitised. The birds recorded in the winter comprised a mixture of residents, short and regional-scale migrants and long distance migrants in unknown proportions. Therefore, the overall winter abundance of each species was expressed as the number of individuals per kilometre per transect totalled across all three counts. Totals (per km) were also calculated for each transect for 'all species' and for 'granivorous species'. In winter, the 'granivorous species' group also included Tree Sparrow *Passer montanus*, (this species was too infrequent in spring to be included in the breeding 'granivorous species' group).

Table 9.1**.** Examples of structural habitat details of the 15 hedgerow transects. Data are derived from the LiDAR digital canopy height model. Tree data are for trees > 3m in height; number of trees refers to individual in-hedge trees; total canopy area includes small groups of trees within some transects.

Transect Name	Length, m	Mean hedge height, m	Total hedge volume, m^3	Number of trees	Mean tree height, m	Total canopy area, m^2
CROSS COMPLIANCE, CC						
Chorleys	658	1.61	717.5	5	7.59	536.0
Madkins	677	1.77	930.8	23	7.33	2622.8
Lampers Den	1131	1.89	2966.1	17	9.51	2051.8
Warren	865	1.70	862.5	12	9.74	2329.0
Ivy Ground	1029	1.78	1984.7	8	7.07	905.5
ENTRY-LEVEL SCHEME, ELS						
Prisnel	1210	1.76	1744.6	22	7.88	2962.5
Shedfield	1119	2.06	3536.2	4	5.20	1422.0
Rimler	944	1.93	1961.8	19	8.42	1706.3
Charlies	1468	1.58	942.0	20	8.93	2040.5
Cow House	1257	1.70	1880.6	21	8.88	2050.0
ENTRY-LEVEL SCHEME EXTRA, ELS-X						
Keepers	1275	1.79	2389.4	25	7.31	2183.3
Ropers	636	1.85	1497.4	6	6.76	391.0
Long Meadow	1803	2.00	4628.0	17	7.50	2683.8
Churchill	857	1.67	1124.1	7	7.10	1129.0
Owl House	1223	1.90	2676.9	18	7.92	2427.3

9.2.4 Bird Use of Winter Food Patches

Bird use of the bird food patches, and relative use of each patch type, was monitored throughout the winter in 2007/08. Counts of birds, identified to species, present in all 20 sown patches were made on each of six visits (October, November, December, early January, late January and February). To minimise the risk of multiple counts of birds moving between patches, all patches were counted on the same day at approximately the same time by five observers each counting four patches. Patches were observed at a distance and eventually flushed by walking around the patch perimeter and then through the patch. The aim of the counts was to record the total number of birds in each patch and counts took *c.* 20 minutes to one hour to complete depending on bird numbers and species composition and the vegetation density of the patch. To obtain a comparison with bird use of cropland, each patch was paired with an equivalent area of crop, in a similar shape and location, in an adjacent field, which was counted immediately after the patch count.

Patch count data were also collected on three visits (December, January, February) in 2008/09 and on six visits (September, October, November, December, January, February) in 2009/10. These additional data are unlikely to change the conclusions drawn from the 2007/08 results but may add information on the influences of patch location and the nature of the surrounding habitat (e.g. length/height of hedge, numbers and heights of trees etc.) on patch use.

9.2.5 Bird Use of Other Prescribed Habitats in Late Winter

As noted in other studies (e.g. Siriwardena *et al.* 2008), bird usage of the food patches was observed to decline in late winter (late January/February, see later). Thus potential bird use of alternative prescribed habitats, i.e. margins and other patch types, was also monitored in January to the beginning of April in 2008. Samples of different margin and patch types, including bird food patches, located chiefly in the ELS-X Treatments, were counted 12 times, approximately weekly, the observer walking at a steady pace and recording all birds seen or flushed from each habitat type. The habitat types, and the total area of each sample, are given in Table 9.2.

Habitat type	Total area recorded, ha	Total no. of birds counted	Birds per 100 m^2 All species	Granivorous species
Bird food EF2 patch	4.84	1144	2.36	1.81
Tussocky margin	1.34	121	0.90	0.72
Pollen & nectar margin	2.29	61	0.27	0.11
Flower EF1 patch	3.64	45	0.12	0.04
Annual cultivation margin	1.28	10	0.08	0.02
Natural regeneration margin	0.70	9	0.13	0.00

Table 9.2 Extent and use of different patch and margin habitat types in late winter (January to first week of April, 2008). EF1 (management of field corners: creation of grass and wildflower patches, 0.5 ha, in field corners that are awkward to reach with machinery) and EF2 (wild bird seed mixture patches: mixtures of annual and biennial small seed-bearing crop species sown in low yielding or awkward patches, 0.25-0.5 ha) are ES habitat designations (Anon 2009).

9.2.6 Tit Breeding Success

A total of 75 tit boxes and 15 sparrow boxes were put up in February 2007. Each sparrow box has three separate compartments allowing for multiple occupation. Thus a total of 120 nest sites were available. All boxes were accessible to both tit species (and also to sparrows). The boxes were supplied at low density to avoid increasing the natural breeding density of tits on the estate, the aim being to encourage some of the birds to use a box instead of a natural hole. Five tit boxes and one sparrow box were put up in each Treatment within in each Block, making a total of 25 tit boxes and five sparrow boxes (total nest sites = 40) per Treatment. Most boxes were attached to hedgerows trees; in the absence of trees, some tit boxes were placed within the structure of the hedge.

The boxes were visited approximately weekly from the beginning of April until July and the following parameters were recorded:

(i) first egg date, i.e. timing of breeding, (ii) clutch size, (iii) number of young alive in the nest at 11 days of age where day of hatching = 0, (iv) mean chick weight (g) at 11 days, excluding runts (runts were defined as chicks too small to be ringed at age 11 days and were relatively uncommon), (v) total live biomass (g) of young in the nest at 11 days (including runts), (vi) number of young fledged, (vii) overall success calculated as the percentage of eggs producing fledged young.

Chicks were weighed to 0.1 g using a spring balance, and were also ringed with a uniquely numbered metal ring of the British ringing scheme (run by the British Trust for

Ornithology, BTO). After the young had fledged, the nest was removed from the box and searched for dead chicks and unhatched eggs.

Tits are well known to have reduced reproductive success in secondary habitats such as parks, gardens and small woods (e.g. Cowie & Hinsley 1987, Hinsley *et al*. 1999, 2008, 2009) but data from farmland are relatively scarce (but see Riddington & Gosler 1995). Farmland, with its relative lack of trees and the exposure of those present in hedgerows and as single trees, may present tits with foraging difficulties. In woodland, they feed their young chiefly on tree-dwelling lepidopteran larvae which can be superabundant for a short period in the spring. Such superabundance is likely to be uncommon or absent in farmland trees and thus the tits might be expected to make greater use of alternative habitats such as the hedgerows themselves and the herbaceous vegetation of the various ES options. This work examined the influence of the habitat around the nest boxes on tit breeding success and assessed the potential for tit species to act as indicators for the presence/quality of certain habitat types. Success was also compared to that typical of pairs breeding in woodland.

9.3 **Statistical analyses**

To account for non-normality and the non-negative integer property of the data, a Poisson modelling approach was used for the breeding, productivity, winter and food patch count data. Where the fit of Poisson models was poor, P values were corrected using the quasipoisson adjustment, all models being run in R 2.10.0 (R Development Core Team 2009).

9.3.1 Breeding Birds

Differences in territory numbers in the breeding season were compared at a whole farm scale using generalized linear models with a Poisson distribution for the bird counts for each transect. Year and Treatment were used as factors plus a log-link function (log transect length) to account for differences in transect lengths. Differences between years were assessed by comparing the model for all years with those in which each year was sequentially removed. When significant differences between years were found, Treatment effects on the differences were examined by comparing models with and without a Year x Treatment interaction. Where Treatment had a significant effect, differences between years were examined separately for each Treatment using transect and year as factors. Treatment effects within years on numbers of territories were examined by comparing models without

and without Treatment as a factor plus the log-link function to account for differences in transect lengths.

Habitat effects on the numbers of territories for three species groups (all species, all granivorous species and all granivorous species without Chaffinch) and for individual species were investigated at a whole farm scale using the 15 transects as replicates. Relationships between the response variables (numbers of territories) and the following habitat variables (corrected for transect length or buffer area as appropriate) were examined using linear regression at **a)** the level of the transects themselves and **b)** a wider landscape-scale using areas of habitat and land-use types within a 100 m buffer around the transects.

a) Transect-level

i) area of all margin and patch habitat within 12 m of the line of the transect

ii) tree canopy area for all trees above 3 m

iii) tree canopy area for tall trees, above 10 m

iv) hedge volume

v) area of all woody vegetation

b) Wider landscape-scale

i) area of all margin and patch habitat

ii) area of all woody vegetation

iii) area of arable land

iv) area of agricultural grassland

v) area of buildings, yards and other man-made structures.

Relationships were examined separately for each habitat variable in each year to identify "best" (and most ecologically relevant) relationships where habitat variables were correlated (see Results) and to remove potentially spurious effects due to outliers. This approach also facilitated the detection of consistency of response and/or trends across years.

9.3.2 Breeding Bird Productivity

Differences in productivity between 2008 and 2010, and Treatment effects on any such differences, were investigated as in a) above using generalized linear models with a Poisson distribution for the numbers of territories and broods for each transect. When examining differences in terms of numbers of broods per territory, the log-link function was omitted and replaced with transect as a factor. Treatment effects within years on numbers of territories and numbers of broods per kilometre were examined as above for the breeding birds. The same approach was used for the numbers of broods per territory, but using 'territories per km' as the log link function instead of transect length. For Bullfinch and Goldfinch in 2008, the data were insufficient to run models including numbers of broods.

Habitat effects on the numbers of territories, numbers of broods per kilometre of transect and numbers of broods per territory were examined using linear regression as described above for the breeding birds.

9.3.3 Winter Birds

Differences in bird numbers in winter were compared as for the breeding birds in a) above using generalized linear models with a Poisson distribution for the bird counts for each transect. Treatment effects within years were also examined as above for the breeding birds.

9.3.4 Bird Use of Winter Food Patches

To examine differences in bird numbers (for all species combined and for individual species) between patch types for each visit, generalized linear models with a Poisson distribution for the bird counts and a log-link function were used, with log (patch area) as an offset to allow for differences in patch area. Patch type (Biennial, Bumblebird, Deluxe) was used as a predictive factor. Models fitting individual slopes for each patch type were compared to those fitting a common slope.

The relationships between individual patch seed production, measured in September (kg of seed per patch, all plant species) and the bird counts (all species combined) from October to February were examined using generalized linear models with a Poisson distribution for the bird counts, patch type as a factor plus seed yield and an interaction between type and seed yield. When the interaction was not significant, models using a single slope and different intercepts for patch type were compared with those fitting a

single slope and intercept. Area was not included as an offset because the response being modelled was the observed response of the birds to the actual seed yield of each patch.

9.3.5 Bird Use of Other Prescribed Habitats in Late Winter

The numbers of birds encountered in all habitats except the bird food patches were small and thus results were expressed as the numbers of birds per habitat type, and numbers per 100 m^2, totalled across all 12 visits. Birds were grouped as 'all species' and 'granivorous species', the latter group being defined as for winter birds.

9.3.6 Tit Breeding Success

The various breeding parameters were averaged for each species in each of the four years and compared qualitatively to breeding performance in woodland (long term study, 1995-present; data combined for three Cambridgeshire woodlands, areas = 157, 136 and 72 ha) for the same years. For the Hillesden data, differences in the parameters between years, and between Treatments within years, were examined using one-way ANOVA with post hoc Tukey tests as required.

Best subsets regression was used to examine the relationships between breeding parameters and the surrounding habitat. All variables (values + 0.01) were transformed to natural logarithms with the exception of overall success (%), which was arcsin-square-root transformed. Transformations were applied to meet the assumptions of linear regression. Breeding performance is strongly influenced by annual variation in climatic variables, thus year, coded as a dummy variable, was included in all regression models in order to detect whether habitat had any consistent effect beyond that attributable to the weather. In addition, best subsets regression was also used to examine each year independently. For all regressions, 'best' models were selected by calculating and comparing Akaike's Information Criterion with adjustment for small sample sizes (AICc). Habitat parameters were obtained from the remote data for two potential foraging distances (radii) of 50 m and 100 m around each nest box. Variables (expressed as totals, maximums, minimums and means as appropriate) included tree canopy height, area and volume, hedgerow length, height, area and volume, ES margin/patch area, a habitat diversity index (Shannon-Weiner, S-W) and a habitat proximity index. The proximity index, adapted from a Fragstats index originally drawn from Gustafson & Parker (1992), was calculated as: index = $\sum (A_i/D_i^2)$ where A_i = area (m^2) of specified habitat (e.g. tree canopy, hedge, ES margin/patch etc.) and D_i = nearest

edge-to-edge distance (m) between the specified habitat and the nest box. Values were summed for all areas of specified habitat whose edges were within the defined distance from the box.

Temperature and rainfall data for March, April and May of each year were obtained from the Met Office (freely available under British Crown copyright 2011, the Met Office: http://www.metoffice.gov.uk/climate/uk/datasets).

9.4 **Results**

9.4.1 Breeding Birds

Breeding season bird abundance expressed as numbers of territories per kilometre and summarised at the scale of the whole study area using the 15 transects as replicates is shown in Table 9.3. The significance of differences between years and of any Treatment effects on such differences are given in Table 9.4. Mean numbers of territories per kilometre for each Treatment are illustrated in Figure 9.1. The general pattern for the species groups and two individual species (Chaffinch and Blue Tit) across the five years was for numbers to increase in the second year (2007), after establishment of the management prescriptions, and then to return to something similar to the baseline level of the first year (2006). Other species (Reed Bunting, Yellowhammer, Blackbird *Turdus merula*, Song Thrush *Turdus philomelos*, Great Tit, Whitethroat *Sylvia communis*, Lesser Whitethroat *Sylvia curruca*) showed relatively little change across the five years whereas Linnet numbers increased in the second year and remained at a higher level thereafter. In contrast, numbers of Dunnocks *Prunella modularis*, Wrens *Troglodytes troglodytes* and especially Robins *Erithacus rubecula* declined in the last two years (2009 and 2010) of the study most probably due to winter weather, especially in 2009/10.

Although significant inter-year differences in numbers of territories were common at the whole-farm scale (Table 9.3 and Table 9.4), significant Treatment effects on differences were found only for Linnet (Figure 9.1). There were more Linnet territories in ELS in 2007 ($F_{1,5} > 100$, $P < 0.001$), 2009 ($F_{1,5} > 100$, $P < 0.001$) and 2010 ($F_{1,5} > 100$, $P < 0.001$) than in 2006, but no significant change in CC. In ELS-X, the increase in numbers was significant between 2006 and 2009 ($F_{1,5} = 4.615$, $P = 0.032$).

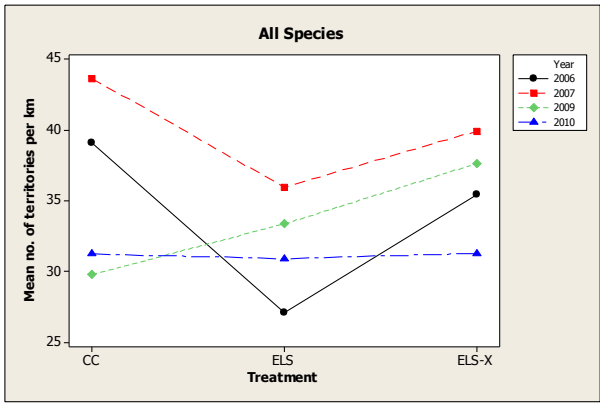
	Breeding season			
Species/group	2006	2007	2009	2010
All species	33.9 (7.51)	39.9 (7.30)	33.6 (5.67)	31.4 (6.21)
Resident species	30.2 (6.43)	36.3 (6.91)	30.1 (4.50)	27.1 (5.43)
Residents without CH	22.9 (5.86)	27.2 (5.67)	23.5 (4.36)	20.5 (4.84)
Granivorous species	13.0 (2.94)	16.0 (3.13)	13.0 (2.39)	12.5 (3.25)
Gran. species without CH	5.7 (2.51)	6.8 (2.13)	6.5 (2.62)	5.9 (2.39)
Chaffinch (CH)	7.3 (1.71)	9.1 (1.92)	6.5 (1.45)	6.6 (1.65)
Linnet	0.7 (1.29)	1.6 (1.52)	2.3 (1.35)	1.8 (0.90)
Reed Bunting	1.2 (1.45)	1.1 (1.15)	1.2 (1.30)	0.9 (0.80)
Yellowhammer	3.8 (1.64)	4.1 (1.27)	3.0 (0.77)	3.2 (1.31)
Blackbird	3.0 (1.55)	3.2 (0.75)	4.5 (1.13)	3.9 (1.71)
Song Thrush	0.6 (0.62)	0.7 (0.60)	1.1 (0.76)	1.0 (0.76)
Great Tit	2.3 (1.23)	2.6 (1.46)	2.5 (0.57)	2.4 (1.19)
Blue Tit	2.8 (1.61)	3.6 (1.44)	2.3 (0.85)	2.3 (1.09)
Dunnock	3.7 (1.43)	4.6 (1.95)	3.0 (1.08)	2.5 (1.03)
Robin	2.8 (1.46)	3.5 (1.38)	1.8 (1.00)	0.8 (0.87)
Wren	2.1 (0.91)	2.2 (1.11)	1.8 (1.02)	1.7 (0.99)
Whitethroat	3.1 (2.35)	2.8 (1.41)	3.1 (1.44)	3.8 (1.31)
Lesser Whitethroat	0.5 (0.54)	0.8 (0.63)	0.4 (0.68)	0.2 (0.45)

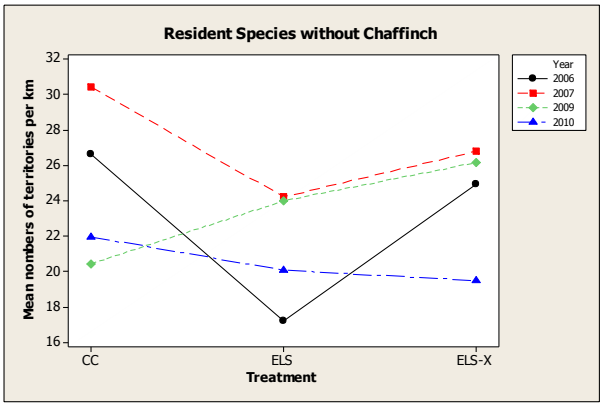
Table 9.3 Breeding season bird abundance at a whole farm scale: mean (SD) numbers of territories per kilometre of transect (n = 15). Winter bird food patches present prior to all breeding seasons except 2006.

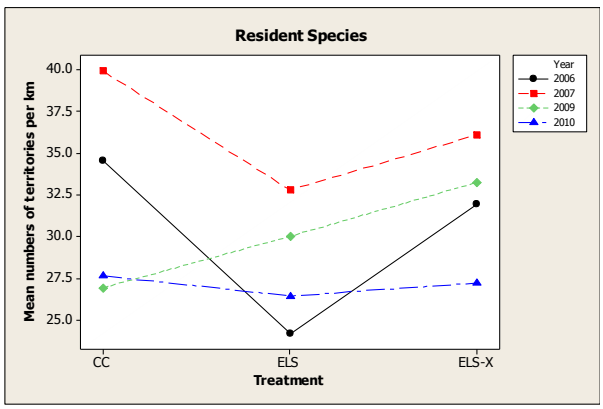
Table 9.4 Significance of inter-year differences in breeding bird territory abundance. Significant Treatment effects on inter-year differences are indicated in bold – see text for details.

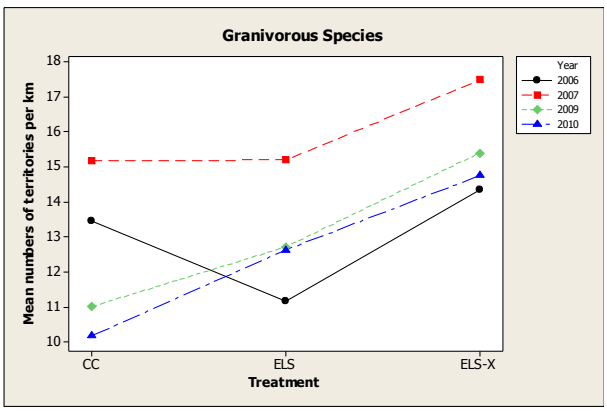
Species/group	2006/07	2006/09	2006/10	2007/09	2007/10	2009/10
All species	$F_{1,55}$ = 8.176 P = 0.004	NS	NS	$F_{1,55}$ = 7.663 P = 0.005	$F_{1,55}$ = 18.129 P < 0.001	NS
Resident species	$F_{1,55}$ = 9.251 P = 0.002	NS	$F_{1,55}$ = 2.973 P = 0.085	$F_{1,55}$ = 8.299 P = 0.003	$F_{1,55}$ = 22.683 P < 0.001	$F_{1,55}$ = 3.553 P = 0.059
Resident species without CH	$F_{1,55}$ = 6.349 P = 0.011	NS	NS	$F_{1,55}$ = 3.324 P = 0.068	$F_{1,55}$ = 15.749 P < 0.001	$F_{1,55}$ = 4.613 P = 0.031
Granivorous spp.	$F_{1,55}$ = 5.307 P = 0.021	NS	NS	$F_{1,55}$ = 5.085 P = 0.024	$F_{1,55}$ = 7.814 P = 0.005	NS
Granivorous spp. without CH	No significant differences between any years					
Chaffinch	$F_{1,55}$ = 2.953 P = 0.086	NS	NS	$F_{1,55}$ = 7.033 P = 0.008	$F_{1,55}$ = 7.033 P = 0.008	NS
Linnet	**$F_{1,55}$ = 5.617 P = 0.018**	**$F_{1,55}$ = 13.604 P < 0.001**	**$F_{1,55}$ = 7.584 P = 0.006**	NS	NS	NS
Reed Bunting	No significant differences between any years					
Yellowhammer	NS	NS	NS	$F_{1,55}$ = 3.099	NS	NS
Blackbird	NS	$F_{1,55}$ = 5.776 P = 0.016	NS	$F_{1,55}$ = 3.989 P = 0.046	NS	NS
Song Thrush	No significant differences between any years					
Great Tit	No significant differences between any years					
Blue Tit	NS	NS	NS	$F_{1,55}$ = 4.482 P = 0.034	$F_{1,55}$ = 3.545 P = 0.060	NS
Dunnock	NS	NS	$F_{1,55}$ = 3.669 P = 0.055	$F_{1,55}$ = 3.989 P = 0.046	$F_{1,55}$ = 9.272 P = 0.002	NS
Robin	NS	NS	$F_{1,55}$ =13.846 P < 0.001	$F_{1,55}$ = 7.648 P = 0.006	$F_{1,55}$ = 25.119 P < 0.001	$F_{1,55}$ = 5.344 P = 0.021
Wren	No significant differences between any years					
Whitethroat	NS	NS	NS P = 0.080	NS	$F_{1,55}$ = 3.072	NS

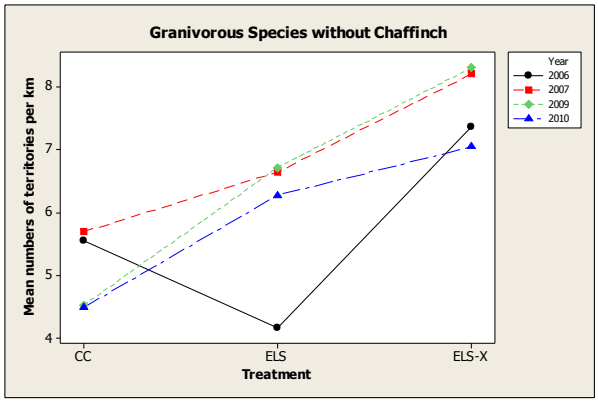
Significant within-year Treatment effects were largely confined to 2006. For 'all species', resident species' and 'resident species without Chaffinch' the greatest numbers of territories occurred in CC, whereas for 'granivorous species' and Linnets, the greatest numbers occurred in ELS-X. For all these results in 2006, fewest territories occurred in ELS (Table 9.5). There were no within-year Treatment effects in 2007. In 2009 and 2010, there were significant effects for 'granivorous species without Chaffinch' and 'granivorous species' respectively and in both cases, the numbers of territories increased in the order of CC < ELS < ELS-X. Despite a lack of statistical significance, in 2009 numbers of territories showed the same increasing trend of CC < ELS < ELS-X for all the species groups and for Linnets, Yellowhammers, Blackbirds and Whitethroats (Figure 9.1). This same trend also occurred for 'granivorous species', 'granivorous species without CH'and Yellowhammers in 2010. Overall, an increasing trend in numbers of territories from CC to ELS to ELS-X was shown in all years beyond the baseline of 2006 for 'granivorous species', 'granivorous species without CH' and Yellowhammers (Figure 9.1).

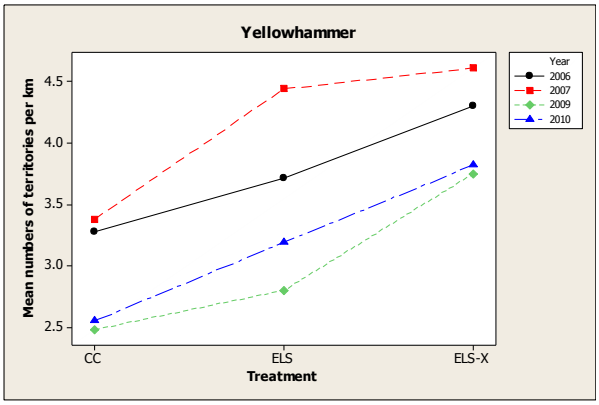


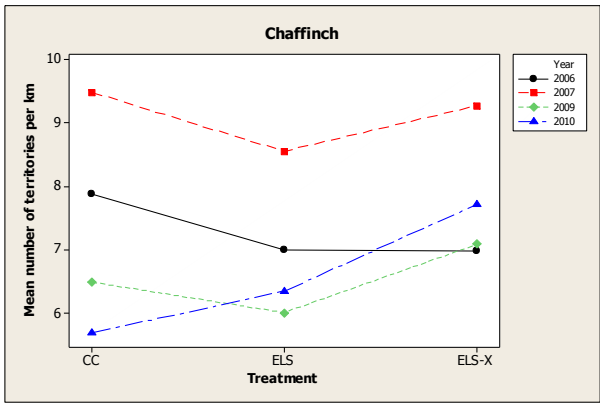


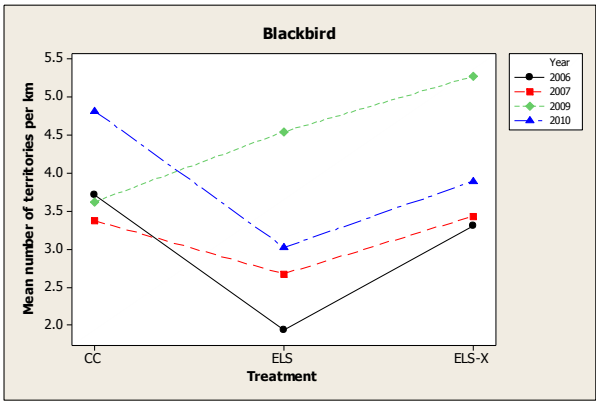


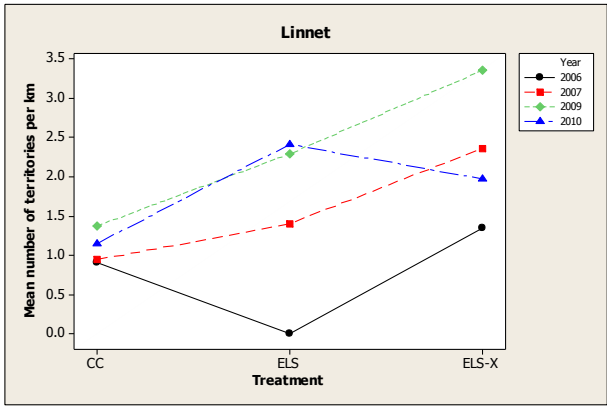


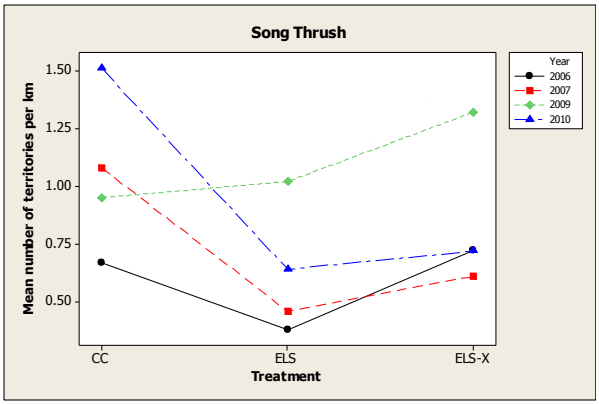

Granivorous Species without Chaffinch
Mean numbers of territories per km
Treatment
CC
ELS
ELS-X
Year
2006
2007
2009
2010

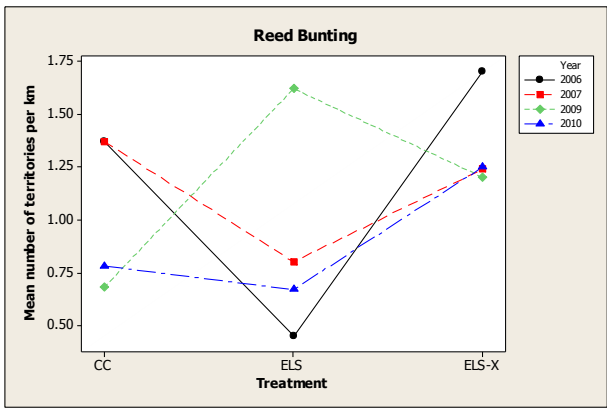

Yellowhammer
Mean numbers of territories per km
Treatment
CC
ELS
ELS-X
Year
2006
2007
2009
2010

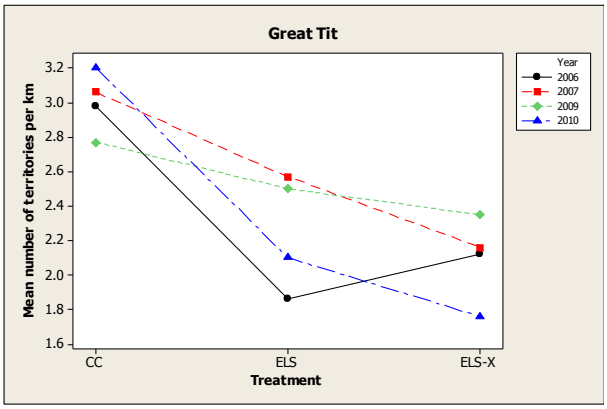

Chaffinch
Mean number of territories per km
Treatment
CC
ELS
ELS-X
Year
2006
2007
2009
2010


Blackbird
Mean number of territories per km
Treatment
CC
ELS
ELS-X
Year
2006
2007
2009
2010


Linnet
Mean number of territories per km
Treatment
CC
ELS
ELS-X
Year
2006
2007
2009
2010


Song Thrush
Mean number of territories per km
Treatment
CC
ELS
ELS-X
Year
2006
2007
2009
2010


Reed Bunting
Mean number of territories per km
Treatment
CC
ELS
ELS-X
Year
2006
2007
2009
2010


Great Tit
Mean number of territories per km
Treatment
CC
ELS
ELS-X
Year
2006
2007
2009
2010

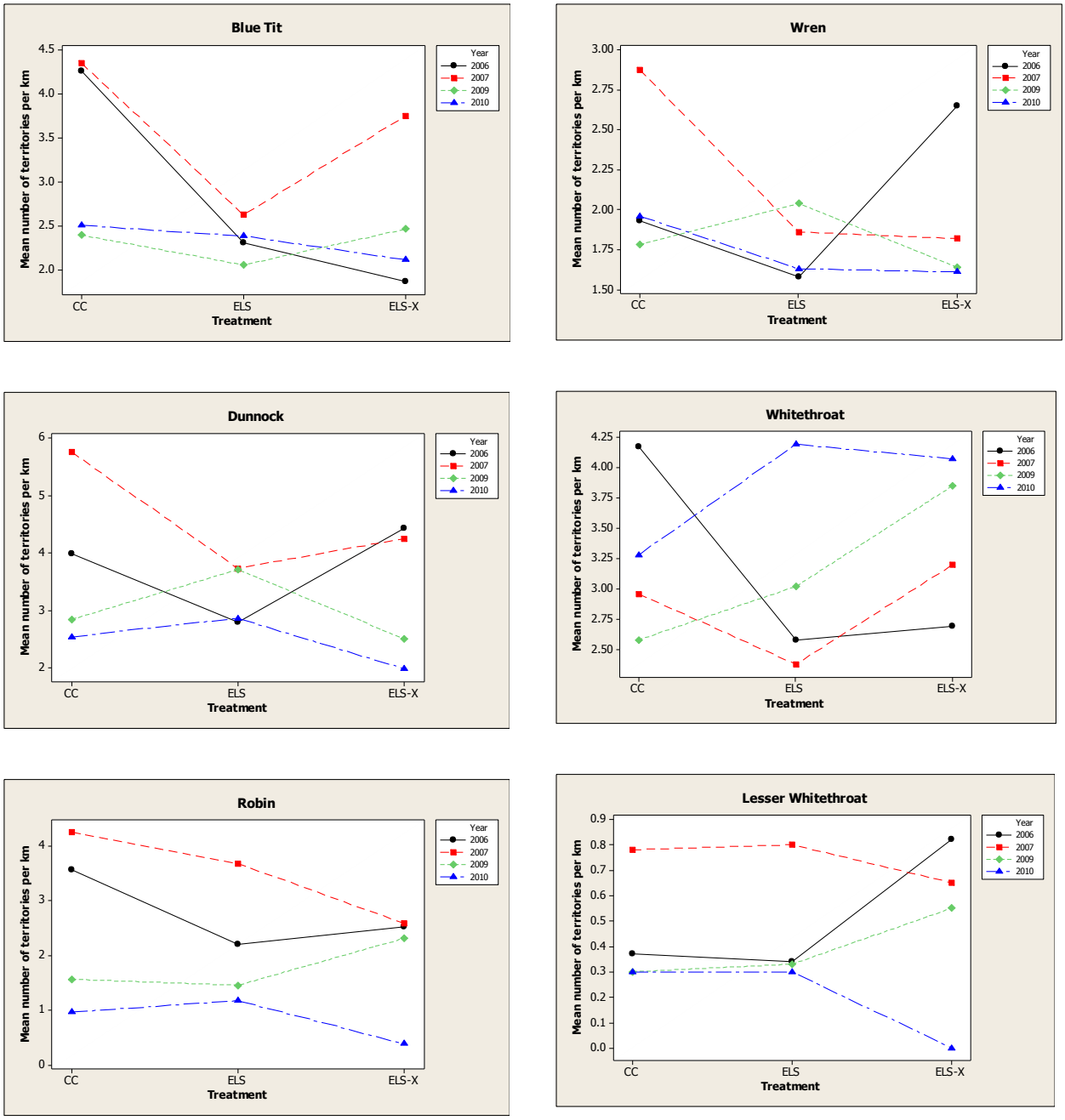


Figure 9.1 Breeding birds. Mean numbers of territories per km of transect for each of the three Treatments for the breeding seasons of 2006 (black), 2007 (red), 2009 (green) and 2010 (blue).

			Numbers of territories, mean ± SD		
Species/group	$F_{12,14}$	P	CC	ELS	ELS-X
2006					
All species	12.81	0.002	39.1 ± 6.72	27.1 ± 3.49	35.4 ± 6.79
Resident species	11.07	0.004	34.5 ± 5.68	24.2 ± 3.50	31.9 ± 5.29
Resident species without CH	12.99	0.002	26.7 ± 4.89	17.2 ± 2.51	24.9 ± 5.12
Granivorous species without CH	4.97	0.083	5.6 ± 1.71	4.2 ± 1.79	7.4 ± 3.08
Linnet	12.18	0.002	0.9 ± 1.53	0	1.3 ± 0.68
2009					
Granivorous species without CH	4.52	0.104	4.5 ± 2.68	6.7 ± 1.12	8.3 ± 2.58
2010					
Granivorous species	4.09	0.129	10.2 ± 3.23	12.6 ± 2.01	14.8 ± 3.04

Table 9.5 Within-year Treatment effects on numbers of territories for the breeding seasons of 2006, 2009 and 2010. There were no significant effects in 2007. Effects significant up to approximately the 10% level are shown (all other results were not significant).

Overall, the presence of a detectable response in the numbers of territories to habitat types increased in the later years of the study, especially in 2009 in relation to the area of margin and patch habitat. In particular, the numbers of territories of 'all species' and 'granivorous species without Chaffinch' increased with the availability of margin and patch habitat in 2009, and that of 'granivorous species' increased in both 2009 and 2010 (Table 9.6 and Table 9.7). This effect was evident at both the transect level and wider landscape scales, but the areas of margin and patch habitat were positively correlated at the two scales (Table 9.8). Although there were no consistent, strong responses to habitat across years, the patterns of those that did occur were generally compatible with the species ecologies, for example Blue Tits favoured trees whereas the seed eaters usually showed negative relationships with trees and woody vegetation in general, and favoured arable land. The negative relationship for Dunnocks with margin and patch habitat at both scales contrasted with this species widespread use of bird food patches in the winter, peaking in December and January (Appendix 12.4, Figure 3d). Initial results indicated positive relationships with buildings for many species in several years at the wider landscape scale, but these were due to the influence of a single outlier and were considered unreliable.

Table 9.6 Transect-level relationships between numbers of territories per kilometre of transect and selected habitat variables within the transect plus a 12 m buffer. Results are given for univariate linear regressions significant up to about P <0.1 (see text) and are expressed at a whole-farm scale using transects (n = 15) as replicates. Note that, despite the difference in methodology, results from the analysis of productivity (2008 and 2010b) are included to facilitate comparisons of changes or trends across years.

		Response variable = number of territories per km			
Habitat variable	Year	Species/group	Direction	R^2	P
Margins & patches	2006	Blue Tit	-	0.39	0.013
(area per km)	2007	All NS			
	2008	Chaffinch	+	0.25	0.055
	2009	All species	+	0.34	0.023
		Gran. species	+	0.42	0.009
		Gran. spp. – CH	+	0.40	0.012
		Whitethroat	+	0.01	0.011
		Yellowhammer	+	0.29	0.022
		Linnet	+	0.27	0.047
		Song Thrush	+	0.25	0.058
		Blackbird	+	0.23	0.070
	2010a	Gran. spp	+	0.37	0.017
		Chaffinch	+	0.39	0.013
		Reed Bunting	+	0.24	0.065
	2010b	Dunnock	-	0.45	0.007
		Reed Bunting	+	0.30	0.035
		Wren	-	0.30	0.036
Trees, > 3 m	2006	Robin	+	0.47	0.005
(canopy area per km)	2007	Blue Tit	+	0.45	0.007
		Chaffinch	+	0.40	0.011
	2008	Robin	+	0.45	0.006
	2009	Gran. spp. – CH	-	0.46	0.006
		Linnet	-	0.50	0.003
		Yellowhammer	-	0.42	0.010
		Wren	-	0.44	0.007
Trees, > 3 m cont.	2010a	Gran. spp. – CH	-	0.40	0.011
		Blue Tit	+	0.51	0.003
		Yellowhammer	-	0.50	0.003
		Reed Bunting	-	0.24	0.066
	2010b	Robin	+	0.78	< 0.001
		Wren	+	0.47	0.005
Tall trees, > 10 m	2006	Chaffinch	+	0.38	0.015
(canopy area per km)		Robin	+	0.27	0.049
		Dunnock	-	0.31	0.030
		Yellowhammer	-	0.25	0.061
	2007	Whitethroat	-	0.29	0.039
	2008	All NS			
	2009	Gran. spp.	-	0.35	0.021
		Gran. spp. – CH	-	0.43	0.008
		Linnet	-	0.65	< 0.001
		Yellowhammer	-	0.32	0.027
	2010a	Gran. spp.	-	0.34	0.024
		Gran. spp. – CH	-	0.41	0.010
		Blue Tit	+	0.35	0.020
		Yellowhammer	-	0.47	0.005
		Wren	-	0.37	0.016
	2010b	Robin	+	0.36	0.019
Hedge volume	2006	Linnet	+	0.30	0.035
(per km)	2007, 2008, 2009 = All NS				
	2010a	Dunnock	+	0.24	0.061
	2010b	Blackbird	-	0.28	0.042
		Linnet	+	0.22	0.079

Table 9.6 (cont.)

Response variable = number of territories per km Habitat variable	Year	Species/group	Direction	R^2	P
All woody veg.	2006, 2008 = All NS				
(area per km)	2007	Chaffinch	+	0.44	0.007
	2007	Linnet	-	0.23	0.071
	2009	Gran. spp. – CH	-	0.36	0.019
		Wren	-	0.27	0.045
		Yellowhammer	-	0.24	0.065
	2010a	Blue Tit	+	0.43	0.008
		Reed Bunting	-	0.24	0.064
	2010b	Robin	-	0.38	0.014

Table 9.7 Wider landscape-level relationships between numbers of territories per kilometre of transect and selected habitat variables within the transect plus a 100 m buffer. Results are given for univariate linear regressions significant up to about P = 0.05 (see text) and are expressed at a whole-farm scale using transects (n = 15) as replicates. Note that, despite the difference in methodology, results from the analysis of productivity (2008 and 2010b) are included to facilitate comparison of trends across years.

		Response variable = number of territories per km			
Habitat variable	Year	Species/group	Direction	R^2	P
Margins & patches	2006	Blue Tit	-	0.29	0.041
(per ha)	2007	Robin	-	0.23	0.072
	2008	Greenfinch	-	0.26	0.052
	2009	All species	+	0.28	0.044
		Gran. species	+	0.48	0.004
		Gran. species – CH	+	0.25	0.056
		Blackbird	+	0.24	0.066
		Yellowhammer	+	0.39	0.014
	2010a	Gran. species	+	0.22	0.076
		Chaffinch	+	0.35	0.021
		Dunnock	-	0.38	0.014
	2010b	Dunnock	-	0.50	0.003
All woody veg.	2006	Robin	+	0.28	0.043
(per ha)	2007, 2009, 2010b = All NS				
	2008	Wren	-	0.24	0.063
		Yellowhammer	-	0.47	0.005
	2010a	Blue Tit	+	0.29	0.040
		Dunnock	-	0.27	0.048
Arable land	2006	Robin	-	0.25	0.057
(per ha)					
	2007	Blue Tit	-	0.28	0.043
	2008	Gran. species	+	0.28	0.043
		Yellowhammer	+	0.47	0.005
		Greenfinch	+	0.28	0.045
		2009	All NS		
	2010a	Dunnock	+	0.51	0.003
		Wren	+	0.25	0.058
	2010b	Dunnock	+	0.37	0.017
Grassland	2006	Blue Tit	+	0.25	0.060
(per ha)		Chaffinch	+	0.27	0.048
		Robin	+	0.38	0.014
	2007	Blue Tit	+	0.34	0.022
	2008	Yellowhammer	-	0.43	0.008
	2009	Gran. species – CH	-	0.43	0.008
		Yellowhammer	-	0.32	0.028
		Linnet	-	0.52	0.002
	2010a	Gran. species – CH	-	0.22	0.075
		Blue Tit	+	0.23	0.072
		Yellowhammer	_	0.26	0.053
	2010b	Robin	+	0.35	0.021
		Wren	+	0.41	0.010
Buildings, yards etc.	All years = NS				
(per ha)					

Transect-level		Landscape-level	
Margin and patch area		Margin and patch (M + P) area	
Tree (> 3 m) canopy area		All woody vegetation area	
Tall tree (> 10 m) canopy area		Arable land area	
Hedge volume		Agricultural grassland area	
All woody vegetation area		Buildings, yards etc. area	
Within transect-level		**Within Landscape-level**	
	P		P
Tree and Tall tree:	0.002	M + P and Arable:	-0.010
Tree and All woody veg:	0.002	Grass and Arable:	-0.001
		All woody veg and Arable:	-0.001
		All woody veg and Grass:	0.005
Between levels			
			P
Transect M + P and Landscape M + P:			0.002
Transect Tree (> 3 m) and Landscape All woody veg:			0.085
Transect Tall tree (> 10 m) and Landscape All woody veg:			0.069

Table 9.8. Correlations between habitat and land-use variables at transect and wider landscape-scales. (Significant correlations only.)

9.4.2 Breeding Bird Productivity

The numbers of breeding territories and breeding productivity, expressed as the mean numbers of broods fledged per kilometre and per territory, are shown in Table 9.9.

	Territories per km		Broods per km		Broods per territory	
Species/group	2008	2010	2008	2010	2008	2010
All species	35.8 (10.9)	35.7 (8.05)	16.2 (9.42)	20.8 (9.80)	0.44 (0.17)	0.57 (0.17)
Resident spp.	32.3 (9.92)	31.1 (6.94)	13.9 (8.82)	16.7 (8.88)	0.41 (0.17)	0.52 (0.17)
Gran. species	15.7 (5.69)	17.2 (3.83)	5.1 (3.15)	7.7 (3.73)	0.31 (0.15)	0.44 (0.15)
Chaffinch	6.7 (1.54)	7.7 (2.05)	2.6 (1.67)	3.4 (2.59)	0.39 (0.24)	0.44 (0.23)
Linnet	1.7 (1.48)	2.5 (1.50)	0.3 (0.41)	0.5 (0.66)	0.15 (0.28)	0.20 (0.30)
Reed Bunting	1.8 (1.27)	1.5 (0.79)	0.4 (0.74)	0.7 (0.94)	0.30 (0.77)	0.37 (0.45)
Yellowhammer	3.5 (1.17)	3.7 (1.05)	0.7 (0.92)	1.8 (1.44)	0.16 (0.22)	0.50 (0.39)
Greenfinch	1.0 (1.48)	0.9 (0.89)	0.8 (1.45)	0.7 (0.92)	0.32 (0.47)	0.47 (0.52)
Bullfinch	0.8 (1.10)	0.6 (0.63)	0.1 (0.30)	0.3 (0.58)	0.10 (0.28)	0.27 (0.46)
Goldfinch	0.2 (0.46)	0.2 (0.54)	0.2 (0.42)	0.3 (0.58)	0.13 (0.35)	0.27 (0.59)
Blackbird	4.4 (2.04)	4.8 (1.57)	2.3 (2.34)	3.3 (2.71)	0.49 (0.43)	0.65 (0.45)
Song Thrush	1.5 (1.09)	1.3 (0.75)	0.7 (0.94)	0.5 (0.86)	0.37 (0.58)	0.36 (0.58)
Dunnock	3.6 (1.10)	3.4 (1.75)	1.5 (1.19)	2.4 (1.88)	0.40 (0.32)	0.73 (0.58)
Robin3.0 (1.09)	1.3 (1.19)	1.9 (1.83)	1.4 (1.66)	0.62 (0.42)	0.93 (0.81)	
Wren 4.0 (1.27)	3.0 (1.02)	2.5 (1.31)	1.3 (1.52)	0.66 (0.32)	0.41 (0.39)	
Whitethroat	3.6 (1.28)	4.7 (1.50)	2.3 (1.17)	4.1 (1.67)	0.70 (0.35)	0.95 (0.42)

Table 9.9 Breeding success in 2008 and 2010 at a whole farm scale: i) mean (SD) numbers of territories per kilometre of transect and ii) mean (SD) numbers of fledged broods per kilometre of transect (n = 15) and per territory. Winter bird food patches present prior to both breeding seasons.

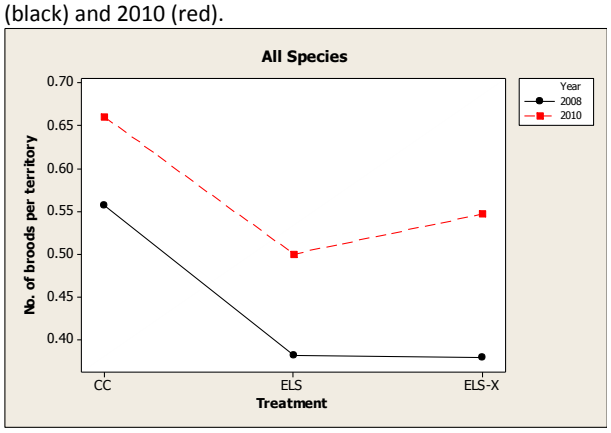
The significance of differences between years is given in Table 9.10. The mean numbers of broods per territory for each Treatment are shown in Figure 9.2. For most species and for the species groups, there was little change in the numbers of territories between 2008 and

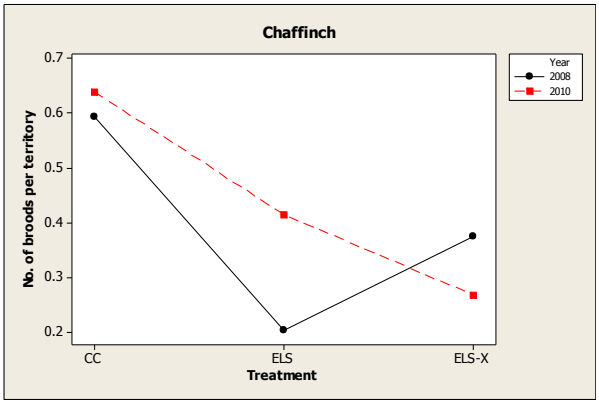
2010. One exception was Linnet which showed a significant increase in numbers of territories in 2010, a result in accordance with the general increase in Linnet territories seen for the breeding bird data in a) above. Robins and Wrens in contrast showed a decrease in 2010, as also detected above for the breeding birds in relation to winter weather, but the change was only significant for Robin. Whitethroats showed an increase, but as summer migrants were not exposed to the conditions which appeared to have affected the former two species (and the difference was not quite significant even at the 10% level, P = 0.111). In terms of broods per kilometre, the breeding season of 2010 appeared to be more productive for most species, the exceptions being Greenfinch, Song Thrush, Robin and Wren. This apparent increase in productivity was also evident at the territory level (broods per territory) for all species except Song Thrush and Wren, although not all these increasing trends were significant (Figure 5.1**,** Table 9.10). For all three measures of productivity, there were no Treatment effects on any of the differences between 2008 and 2010.

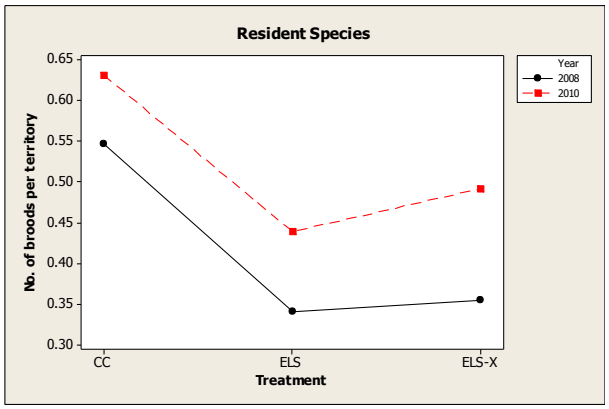
Species/group	Territories per km 2008/10	Broods per km 2008/10	Broods per territory 2008/10
All species	NS	NS	$F_{1,15} = 8.631$ P = 0.011
Resident species	NS	NS	$F_{1,15} = 5.303$ P = 0.037
Granivorous species	NS	$F_{1,27} = 9.764$ P = 0.019	$F_{1,15} = 11.362$ P = 0.005
Chaffinch	NS	NS	NS
Linnet	$F_{1,27} = 3.193$ P = 0.074	NS	NS
Reed Bunting	NS	NS	NS
Yellowhammer	NS	$F_{1,27} = 7.366$ P = 0.007	$F_{1,15} = 11.447$ P = 0.004
Greenfinch	NS	NS	NS
Bullfinch	NS	NS	NS
Goldfinch	NS	NS	NS
Blackbird	NS	NS	NS
Song Thrush	NS	NS	NS
Dunnock	NS	$F_{1,27} = 2.991$ P = 0.084	$F_{1,15} = 4.493$ P = 0.052
Robin	$F_{1,27} = 11.880$ P < 0.001	NS	NS
Wren	NS	$F_{1,27} = 6.796$ P = 0.009	NS
Whitethroat	NS	$F_{1,27} = 7.458$ P = 0.006	$F_{1,15} = 4.646$ P = 0.049

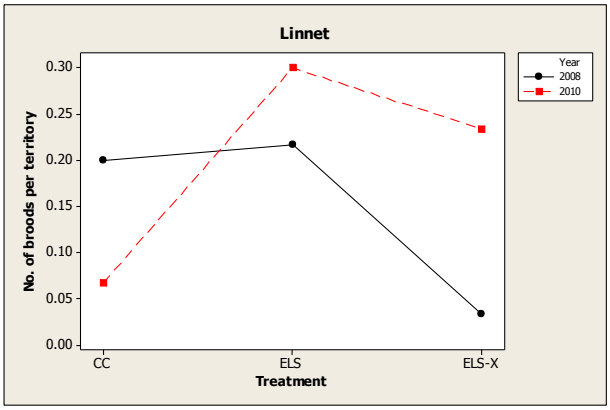
Table 9.10**.** Significance of differences in productivity between 2008 and 2010. There were no Treatment effects on any of the inter-year differences.

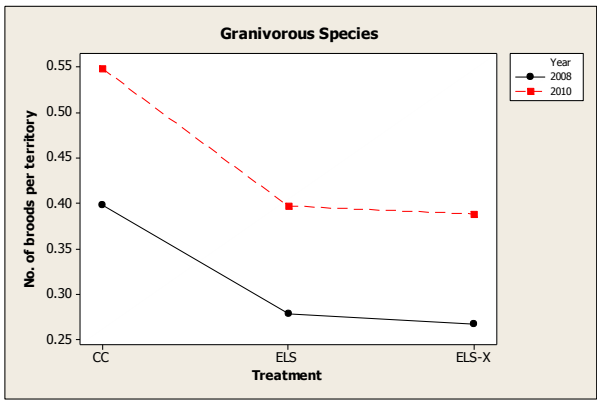
Figure 9.2. Breeding bird productivity. Mean numbers of broods per territory for each of the three Treatments in 2008 (black) and 2010 (red).

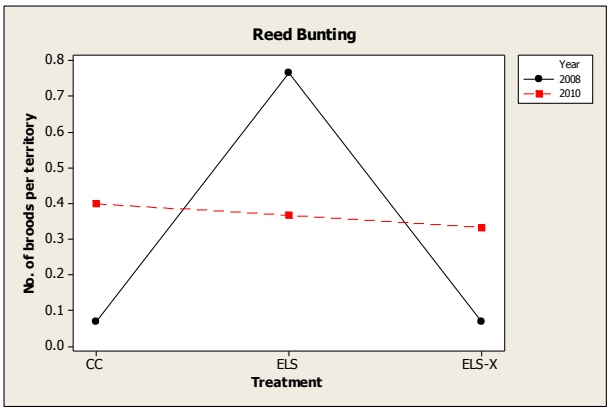


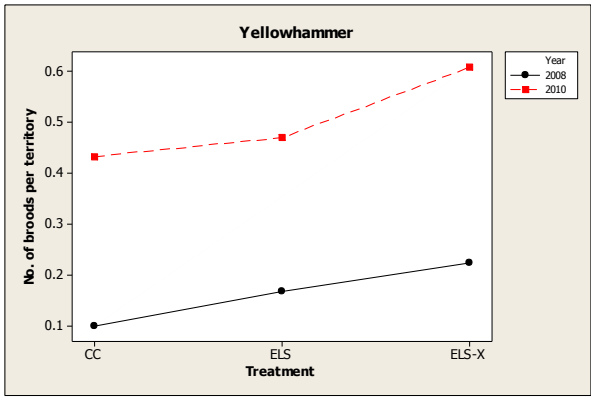


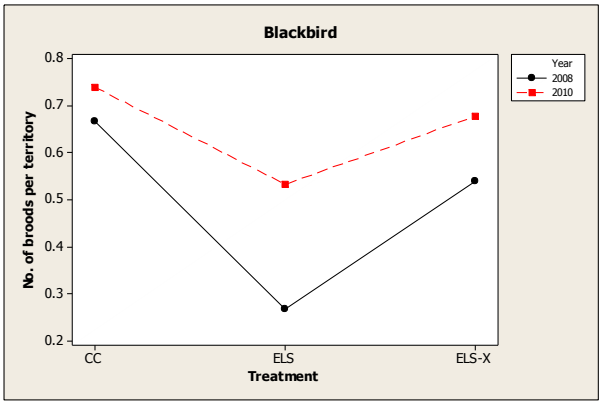


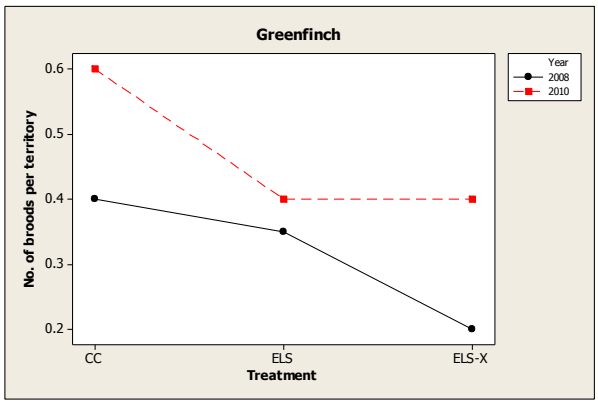


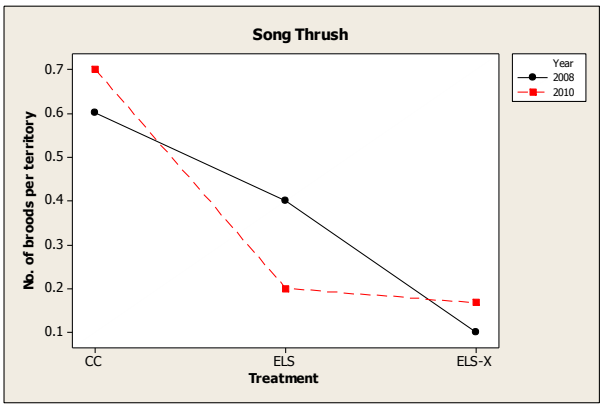





Yellowhammer
No. of broods per territory
0.6
0.5
0.4
0.3
0.2
0.1
CC
ELS
ELS-X
Treatment
Year
2008
2010


Blackbird
No. of broods per territory
0.8
0.7
0.6
0.5
0.4
0.3
0.2
CC
ELS
ELS-X
Treatment
Year
2008
2010


Greenfinch
No. of broods per territory
0.6
0.5
0.4
0.3
0.2
CC
ELS
ELS-X
Treatment
Year
2008
2010


Song Thrush
No. of broods per territory
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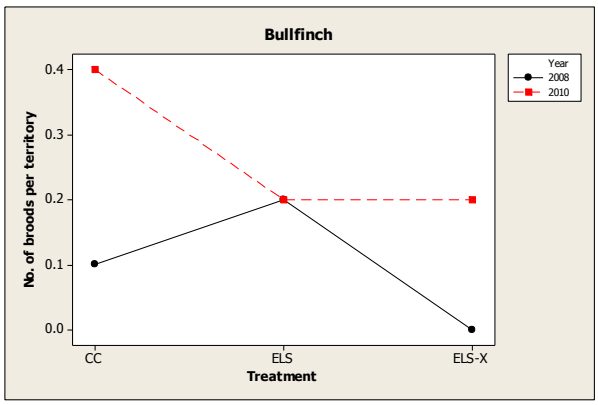
0.6
0.5
0.4
0.3
0.2
0.1
CC
ELS
ELS-X
Treatment
Year
2008
2010


Bullfinch
No. of broods per territory
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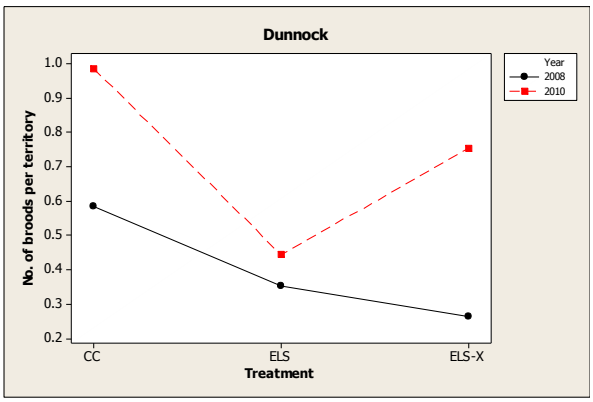
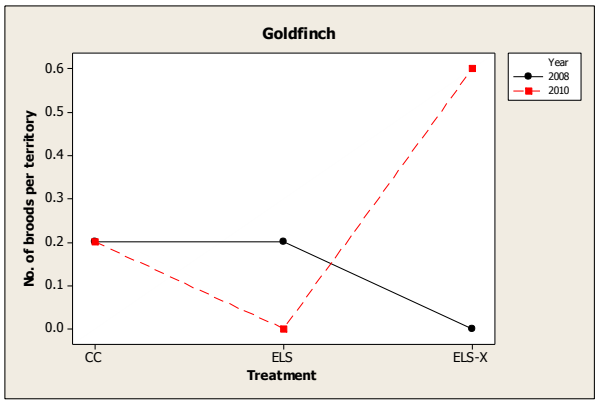
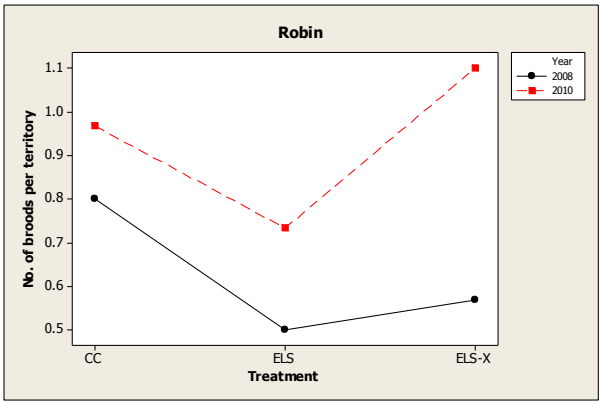
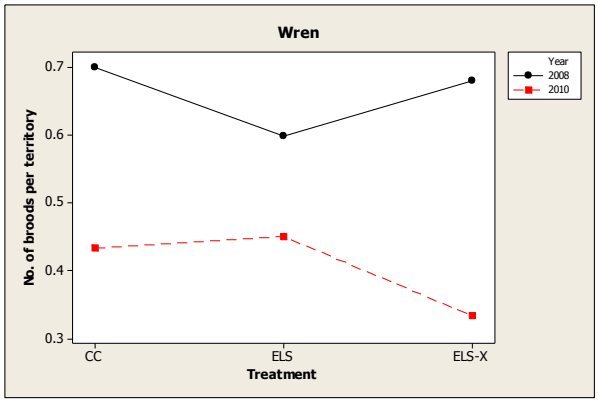
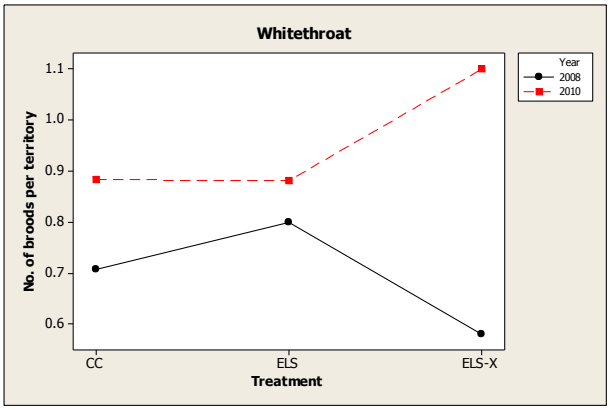
0.3
0.2
0.1
0.0
CC
ELS
ELS-X
Treatment
Year
2008
2010


Dunnock
No. of broods per territory
1.0
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
CC
ELS
ELS-X
Treatment
Year
2008
2010


Goldfinch
No. of broods per territory
0.6
0.5
0.4
0.3
0.2
0.1
0.0
CC
ELS
ELS-X
Treatment
Year
2008
2010


Robin
No. of broods per territory
1.1
1.0
0.9
0.8
0.7
0.6
0.5
CC
ELS
ELS-X
Treatment
Year
2008
2010


Wren
No. of broods per territory
0.7
0.6
0.5
0.4
0.3
CC
ELS
ELS-X
Treatment
Year
2008
2010


Whitethroat
No. of broods per territory
1.1
1.0
0.9
0.8
0.7
0.6
CC
ELS
ELS-X
Treatment
Year
2008
2010

Within years, there were no Treatment effects on the numbers of territories per kilometre in 2008, and only marginally for Greenfinch ($\chi_{12,14}$ = 4.92, P = 0.086) in 2010, with the greatest numbers in CC (mean ± SD: CC = 1.7 ± 0.8, ELS = 0.4 ± 0.5, ELS-X = 0.5 ± 0.8). For the numbers of broods per kilometre, there were significant Treatment effects for Blackbird in 2008 ($\chi_{12,14}$ = 8.49, P = 0.014) and Chaffinch in 2010 ($\chi_{12,14}$ = 5.18, P = 0.075) and in both cases numbers were again greatest in CC (Blackbird: CC = 4.0 ± 3.1, ELS = 1.0 ± 1.0, ELS-X = 1.8 ± 1.6; Chaffinch: CC = 5.4 ± 3.9, ELS = 2.8 ± 0.5, ELS-X = 2.1 ± 0.5). For Goldfinches in 2010, most broods were recorded in ELS-X ($\chi_{12,14}$ = 5.82, P = 0.055; CC = 0.2 ± 0.9, ELS = 0, ELS-X = 0.8 ± 1.1), but the result was likely to be unreliable because the data were sparse (five broods in three transects). There were no significant Treatment effects on the numbers of broods per territory.

There were few strong relationships between productivity (broods per kilometre and broods per territory) and habitat types at either the transect or wider landscape scale (Table 9.11 and Table 9.12). Margin and patch habitat appeared to have little influence, the only relationships significant at less than P = 0.05 being for Chaffinch and both were negative. This negative response for Chaffinch was in contrast to the positive relationship with margin and patch habitat for the numbers of Chaffinch territories in 2008 (Table 9.6). Overall, granivorous species, and Whitethroats, which are generally associated with open farmland showed negative responses to trees and related variables (e.g. all woody vegetation) and positive ones to arable land. As with the results for the numbers of territories (Table 9.6), positive relationships with buildings for several species were due to a single outlier. However, such positive relationships for the numbers of broods per kilometre for Blackbirds and Dunnocks did not rely entirely on this outlier, but the result was only strong (P = 0.001) for Blackbirds (Table 9.12).

Table 9.11**.** Transect-level relationships between a) number of broods per kilometre of transect and b) number of broods per territory per transect and selected habitat variables within the transect plus a 12 m buffer. Results are given for univariate linear regressions significant up to about P = 0.05 (see text) and are expressed at a whole-farm scale using transects (n = 15) as replicates.

		Response variable: broods per km & broods per territory			
Habitat variable	Year	Species/group	Direction	R^2	P
a) BROODS PER KILOMETRE					
Margins & patches (area per km)	2008, 2010	= All NS			
Trees, > 3 m (canopy area per km)	2008	Yellowhammer	-	0.23	0.070
		Whitethroat	-	0.38	0.015
	2010	Yellowhammer	-	0.34	0.023
		Robin	+	0.61	0.001
Tall trees, > 10 m (canopy area per km)	2008	Gran. species – CH	-	0.24	0.061
		Yellowhammer	-	0.24	0.063
		Whitethroat	-	0.57	0.001
	2010	Robin	+	0.25	0.058
Hedge volume (per km)	2008	Reed Bunting	-	0.24	0.065
	2010	Gran. species	-	0.22	0.074
Total woody veg. (area per km)	2008	All NS			
	2010	Yellowhammer	-	0.48	0.004
b) BROODS PER TERRITORY					
Margins & patches (area per km)	2008	All NS			
	2010	Chaffinch	-	0.38	0.014
Trees, > 3 m (canopy area per km)	2008	Yellowhammer	-	0.22	0.080
		Whitethroat	-	0.31	0.030
	2010	All NS			
Tall trees, > 10 m (canopy area per km)	2008	Robin	-	0.28	0.041
		Whitethroat	-	0.26	0.055
	2010	Blackbird	-	0.29	0.039
Hedge volume (per km)	2008	All species	-	0.22	0.078
		Gran. species	-	0.24	0.064
		Chaffinch	-	0.23	0.068
		Wren	-	0.23	0.065
	2010	Gran. species	-	0.42	0.009
		Gran. spp. – CH	-	0.32	0.028
		Linnet	-	0.22	0.075
Total woody veg. (area per km)	2008	Wren	-	0.31	0.030
	2010	Yellowhammer	-	0.31	0.028

Table 9.12 Wider landscape-level relationships between a) number of broods per kilometre of transect and b) number of broods per territory per transect and selected habitat variables within the transect plus a 100 m buffer. Results are given for univariate linear regressions significant up to about P = 0.05 (see text) and are expressed at a whole-farm scale using transects (n = 15) as replicates.

		Response variable: broods per km & broods per territory			
Habitat variable	Year	Species/group	Direction	R^2	P
a) BROODS PER KILOMETRE					
Margins & patches (per ha)	2008	Gran. spp. – CH	-	0.23	0.073
		Dunnock	-	0.26	0.054
		Robin	-	0.23	0.070
	2010	All NS			
All woody veg. (per ha)	2008	Gran. species	-	0.22	0.077
		Gran. spp. – CH	-	0.31	0.030
		Linnet	-	0.29	0.037
		Dunnock	-	0.32	0.027
	2010	All NS			
Arable land (per ha)	2008	Gran. species	+	0.24	0.062
		Gran. spp. – CH	+	0.46	0.006
		Dunnock	+	0.25	0.056
		Whitethroat	+	0.28	0.045
	2010	All NS			
Grassland (per ha)	2008	Yellowhammer	-	0.23	0.068
	2010	Robin	+	0.49	0.004
Buildings, yards etc. (per ha)	2008	Blackbird	+	0.59	0.001
		Dunnock	+	0.22	0.078
	2010	All NS			
b) BROODS PER TERRITORY					
Margins & patches (per ha)	2008	All NS			
	2010	Chaffinch	-	0.40	0.011
All woody veg. (per ha)	2008	Gran. species	-	0.24	0.061
		Gran. spp. – CH	-	0.48	0.004
	2010	All NS			
Arable land (per ha)	2008	Gran. spp. – CH	+	0.32	0.028
	2010	All NS			
Grassland (per ha)	2008, 2010 = All NS				
Buildings, yards etc. (per ha)	2008, 2010 = All NS				

9.4.3 Winter Birds

Winter bird abundance expressed as the numbers of birds per kilometre and summarised at the scale of the whole study area using the 15 transects as replicates is shown in Table 9.13. The significance of differences between winters and of Treatment effects on such differences are given in Table 9.14. Significance of inter-year differences in winter bird abundance. Significant Treatment effects on inter-year differences are indicated in bold – see text for details.Mean numbers of birds per kilometre for each Treatment are illustrated in Figure 9.3. The general pattern across the five years for the groups 'all species' and 'granivorous species' was for numbers to increase in the second year (2007), after establishment of the management prescriptions, and then to remain at a higher level thereafter. This pattern was largely driven by the individual responses of Chaffinch, Linnet, Great Tit and Blue Tit, and to a lesser extent, Yellowhammer (Table 9.13). Numbers of Reed Buntings fluctuated and those of Tree Sparrows remained generally low. Blackbird numbers remained similar throughout whereas Song Thrushes, Robins and Wrens increased in the first winter after the management prescriptions were established but then declined below the numbers present in the baseline winter. Dunnocks showed a similar pattern, but although numbers declined in the last two winters, they remained above baseline.

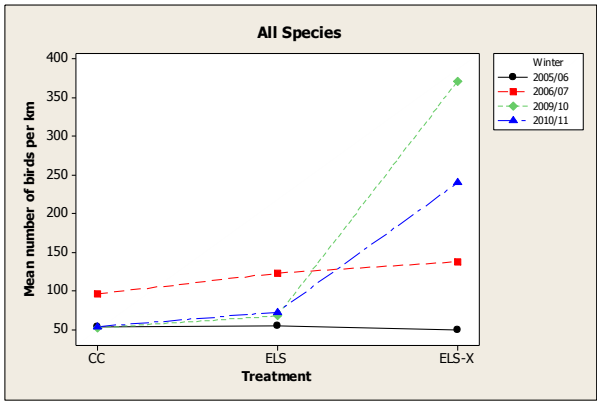
Table 9.13 Winter bird abundance at a whole farm-scale: mean (SD) numbers of birds per kilometre of transect (n = 15). Winter bird food patches present in all winters except 2005/06.

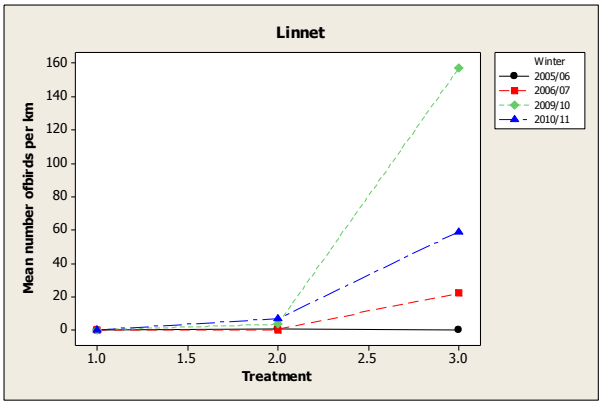
	Winter			
Species/group	2005/06	2006/07	2009/10	2010/11
All species	52.5 (14.94)	118.5 (37.12)	138.3 (230.2)	110.2 (85.00)
Granivorous species	9.9 (5.33)	51.0 (33.41)	97.7 (228.5)	70.6 (78.50
Chaffinch	8.4 (4.23)	27.7 (10.86)	41.1 (78.9)	40.8 (47.10)
Linnet	0.2 (0.56)	7.4 (24.39)	53.5 (149.9)	21.8 (56.60)
Reed Bunting	0.0	3.4 (4.69)	0.2 (0.2)	3.8 (8.44)
Yellowhammer	1.3 (1.29)	11.8 (15.77)	2.8 (5.020	4.0 (7.57)
Tree Sparrow	0.1 (0.29)	0.7 (1.71)	0.2 (0.47)	0.1 (0.30)
Blackbird	10.0 (4.00)	10.8 (4.71)	9.2 (3.48)	8.9 (4.34)
Song Thrush	4.6 (4.65)	8.0 (4.15)	2.8 (2.33)	1.4 (1.77)
Great Tit	2.7 (2.76)	5.3 (4.17)	6.4 (3.18)	5.7 (4.68)
Blue Tit	7.3 (5.31)	13.1 (7.17)	10.1 (4.34)	11.5 (6.26)
Dunnock	4.4 (2.54)	13.0 (6.33)	5.1 (2.94)	5.2 (3.38)
Robin	7.7 (4.30)	8.2 (5.30)	4.0 (2.00)	2.6 (1.42)
Wren	4.4 (3.64)	5.6 (2.44)	3.0 (1.65)	1.9 (1.18)

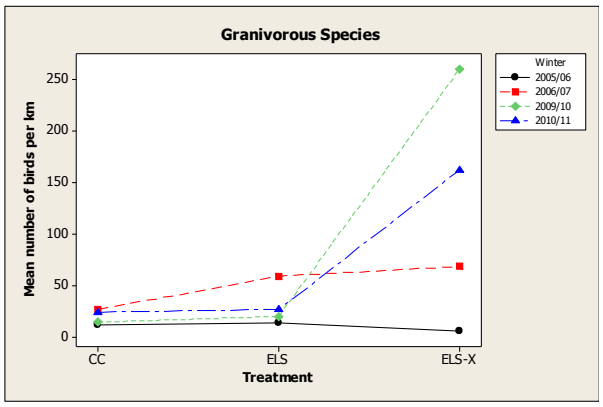
Table 9.14. Significance of inter-year differences in winter bird abundance. Significant Treatment effects on inter-year differences are indicated in bold – see text for details.

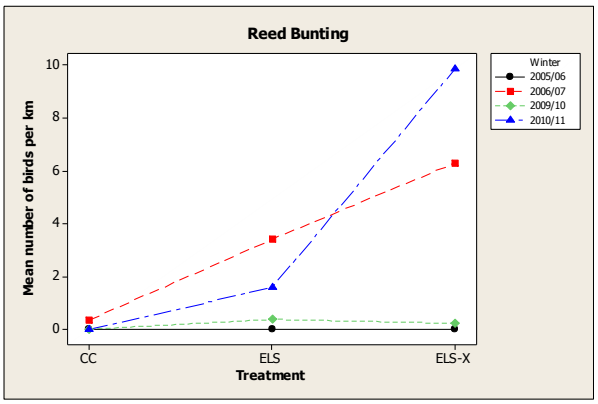
Species/group	2005/06 2006/07	2005/06 2009/10	2005/06 2010/11	2006/07 2009/10	2006/07 2010/11	2009/10 2010/11
All species	$F_{1,55}$ = 5.471 P = 0.023	**$F_{1,55}$ = 10.856** **P = 0.002**	**$F_{1,55}$ = 4.831** **P = 0.032**	NS	NS	NS
Granivorous spp.	**$F_{1,55}$ = 5.548** **P = 0.022**	$F_{1,55}$ = 19.279 P < 0.001	**$F_{1,55}$ = 10.523** **P = 0.002**	**$F_{1,55}$ = 4.748** **P = 0.034**	NS	NS
Chaffinch	**$F_{1,55}$ = 5.189** **P = 0.027**	**$F_{1,55}$ = 13.320** **P < 0.001**	**$F_{1,55}$ = 9.930** **P = 0.003**	NS	NS	NS
Linnet	NS	$F_{1,55}$ = 22.048 P < 0.001	$F_{1,55}$ = 8.481 P = 0.005	$F_{1,55}$ = 15.516 P < 0.001	$F_{1,55}$ = 3.970 P = 0.051	$F_{1,55}$ = 4.397 P = 0.041
Reed Bunting	$F_{1,55}$ = 31.720 P < 0.001	NS	$F_{1,55}$ = 38.517 P < 0.001	$F_{1,55}$ = 23.359 P < 0.001	NS	$F_{1,55}$ = 29.662 P < 0.001
Yellowhammer	$F_{1,55}$ = 14.868 P < 0.001	NS	$F_{1,55}$ = 2.954 P = 0.091	$F_{1,55}$ = 8.948 P = 0.004	$F_{1,55}$ = 5.075 P = 0.028	NS
Tree Sparrow	$F_{1,55}$ = 10.124 P = 0.001	NS	NS	$F_{1,55}$ = 7.723 P = 0.005	$F_{1,55}$ = 13.447 P < 0.001	NS
Blackbird	No significant differences between any years					
Song Thrush	$F_{1,55}$ = 5.393 P = 0.024	NS	$F_{1,55}$ = 9.132 P = 0.004	$F_{1,55}$ = 14.634 P < 0.001	$F_{1,55}$ = 27.602 P < 0.001	NS
Great Tit	**$F_{1,55}$ = 5.996** **P = 0.018**	**$F_{1,55}$ = 7.482** **P = 0.008**	$F_{1,55}$ = 5.793 P = 0.020	NS	NS	NS
Blue Tit	$F_{1,55}$ = 10.011 P = 0.003	$F_{1,55}$ = 3.258 P = 0.077	$F_{1,55}$ = 6.439 P = 0.014	NS	NS	NS
Dunnock	$F_{1,55}$ = 39.808 P < 0.001	NS	NS	$F_{1,55}$ = 32.916 P < 0.001	$F_{1,55}$ = 25.902 P < 0.001	NS
Robin	NS	$F_{1,55}$ = 18.729 P < 0.001	$F_{1,55}$ = 43.719 P < 0.001	$F_{1,55}$ = 22.665 P < 0.001	$F_{1,55}$ = 49.519 P < 0.001	$F_{1,55}$ = 5.481 P = 0.019
Wren	NS	$F_{1,55}$ = 6.076 P = 0.014	$F_{1,55}$ = 20.339 P < 0.001	$F_{1,55}$ = 16.549 P < 0.001	$F_{1,55}$ = 37.010 P < 0.001	**$F_{1,55}$ = 4.304** **P = 0.038**

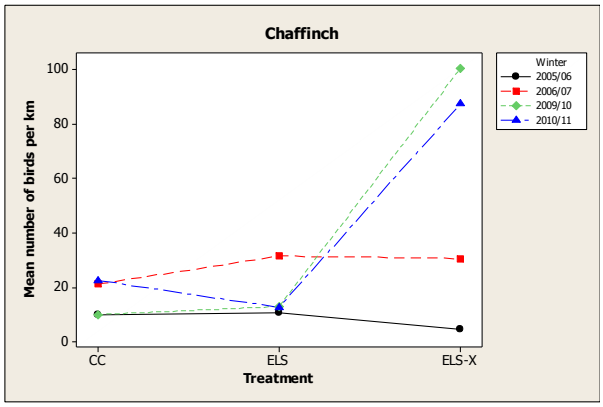
Figure 9.3 Winter birds. Mean numbers of birds per km of transect for each of the three Treatments in the winters of 2005/06 (black), 2006/07 (red), 2009/10 (green) and 2010/11 (blue).

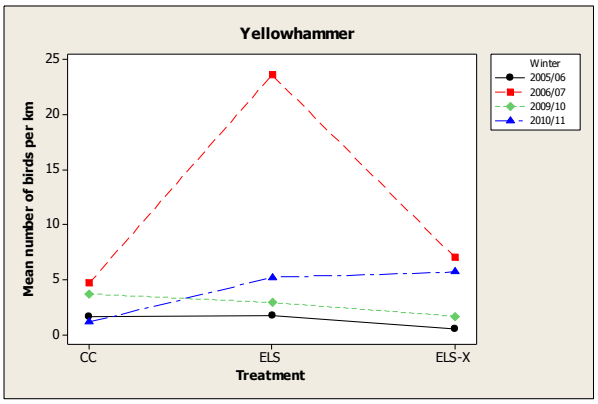


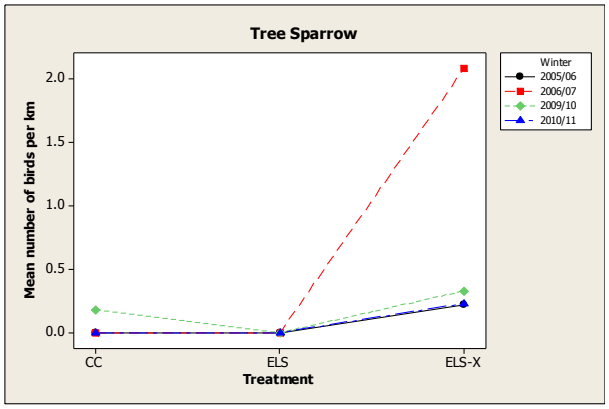


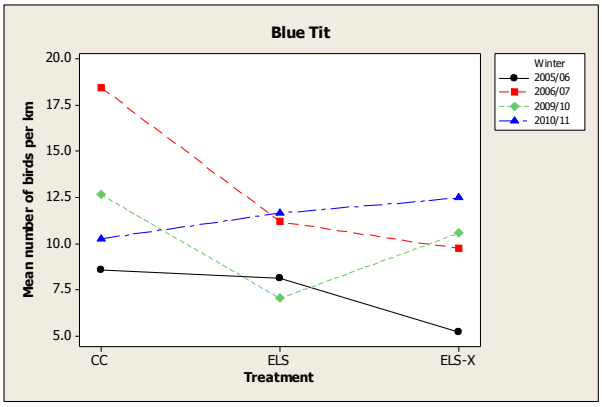


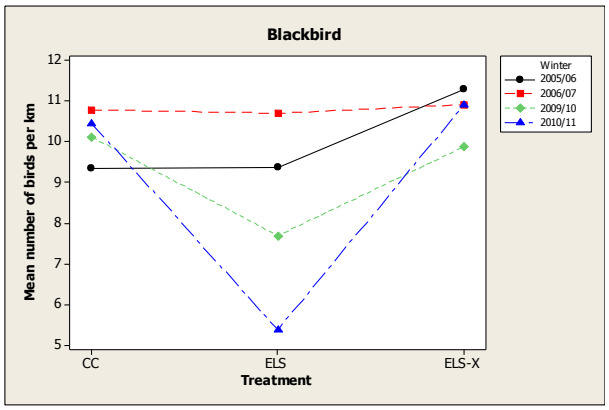


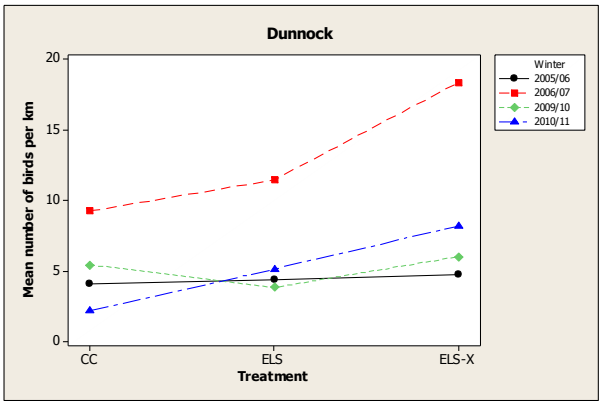


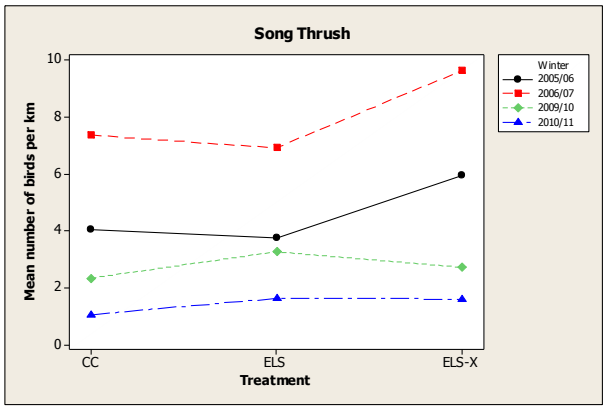


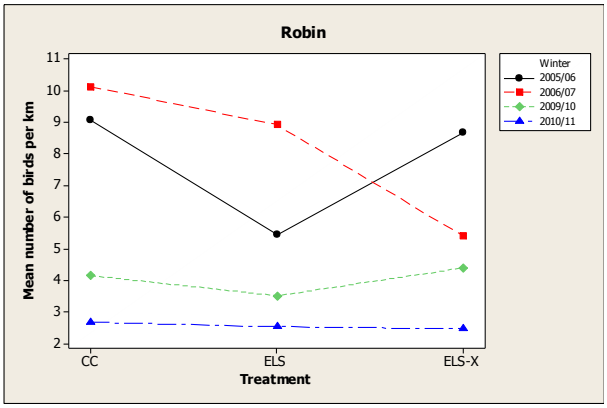

Tree Sparrow
Mean number of birds per km
2.0
1.5
1.0
0.5
0.0
CC
ELS
ELS-X
Treatment
Winter
2005/06
2006/07
2009/10
2010/11

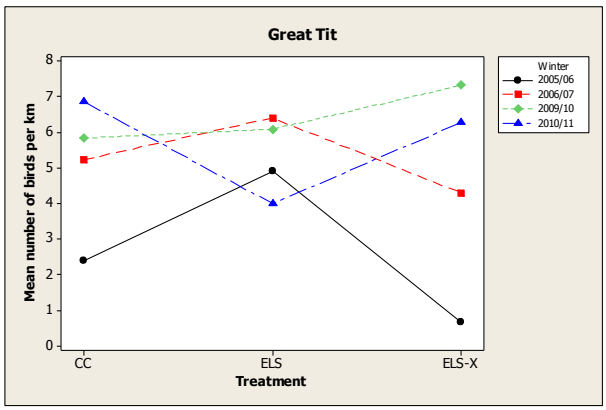

Blue Tit
Mean number of birds per km
20.0
17.5
15.0
12.5
10.0
7.5
5.0
CC
ELS
ELS-X
Treatment
Winter
2005/06
2006/07
2009/10
2010/11

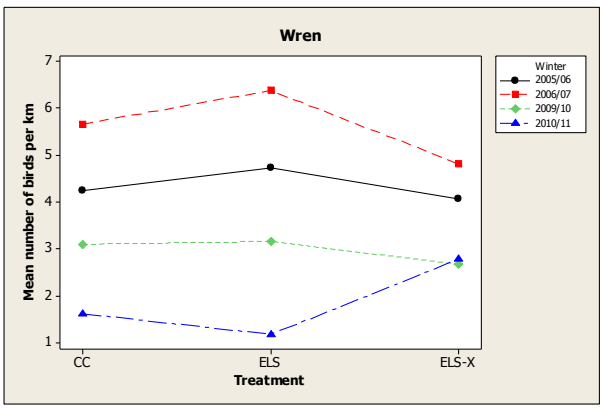

Blackbird
Mean number of birds per km
12
11
10
9
8
7
6
5
CC
ELS
ELS-X
Treatment
Winter
2005/06
2006/07
2009/10
2010/11


Dunnock
Mean number of birds per km
20
15
10
5
0
CC
ELS
ELS-X
Treatment
Winter
2005/06
2006/07
2009/10
2010/11


Song Thrush
Mean number of birds per km
10
8
6
4
2
0
CC
ELS
ELS-X
Treatment
Winter
2005/06
2006/07
2009/10
2010/11


Robin
Mean number of birds per km
11
10
9
8
7
6
5
4
3
2
CC
ELS
ELS-X
Treatment
Winter
2005/06
2006/07
2009/10
2010/11


Great Tit
Mean number of birds per km
8
7
6
5
4
3
2
1
0
CC
ELS
ELS-X
Treatment
Winter
2005/06
2006/07
2009/10
2010/11


Wren
Mean number of birds per km
7
6
5
4
3
2
1
CC
ELS
ELS-X
Treatment
Winter
2005/06
2006/07
2009/10
2010/11

Significant inter-year differences in winter bird abundance were again common at the whole-farm scale (Table 9.14), but there were more Treatment effects, summarised in Table 9.15, on the differences than for the breeding birds. For 'all species' and Chaffinch, there were more birds in ELS-X in 2009/10 and 2010/11 than in 2005/06, and for Chaffinch, more birds in all three Treatments in 2006/07 than in 2005/06. There were also more birds in all three Treatments for the group of 'granivorous species' in 2006/07 than in 2005/06, more in ELS-X in 2010/11 than in 2005/06 and in 2009/10 than in 2006/07, but less in ELS in 2009/10 than in 2006/07. For Great Tits, there were more birds in CC and ELS-X in both 2006/07 and 2009/10 than in 2005/06, and more Wrens in ELS in 2009/10 than in 2010/11 (Table 9.15, Figure 9.3).

Table 9.15 Significant Treatment effects on inter-year differences in winter bird abundance. Numbers were greater in the second winter of the comparison unless indicated as negative by –ve.

Species/group	2005/06 2006/07	2005/06 2009/10	2005/06 2010/11	2006/07 2009/10	2006/07 2010/11	2009/10 2010/11
All species	-	**ELS-X** $F_{1,5}$ = 49.347 P = 0.002	**ELS-X** $F_{1,5}$ = 553.85 P < 0.001	-	-	-
Granivorous spp.	**CC** $F_{1,5}$ = 17.009 P = 0.015	-	**ELS-X** $F_{1,5}$ = 168.51 P < 0.001	**ELS -ve** $F_{1,5}$ = 13.303 P = 0.022	-	-
	ELS $F_{1,5}$ = 36.161 P = 0.004	-	-	**ELS-X** $F_{1,5}$ = 7.288 P = 0.054	-	-
	ELS-X $F_{1,5}$ = 92.740 P < 0.001	-	-	-	-	-
Chaffinch	**CC** $F_{1,5}$ = 8.972 P = 0.040	**ELS-X** $F_{1,5}$ = 61.325 P = 0.001	**ELS-X** $F_{1,5}$ = 275.94 P < 0.001	-	-	-
	ELS $F_{1,5}$ = 16.892 P = 0.015	-	-	-	-	-
	ELS-X $F_{1,5}$ = 52.849 P = 0.002	-	-	-	-	-
Great Tit	**CC** $F_{1,5}$ = 25.687 P = 0.007	**CC** $F_{1,5}$ = 8.447 P = 0.044	-	-	-	-
	ELS-X $F_{1,5}$ = 21.212 P = 0.010	**ELS-X** $F_{1,5}$ = 5.682 P = 0.076	-	-	-	-
Wren	-	-	-	-	-	**ELS -ve** $F_{1,5}$ = 9.177 P = 0.039

Within-year Treatment effects increased across the five years of the study due to the presence of the bird food patches in ELS and ELS-X. These trends were most prominent for 'all species', granivorous species', Chaffinch, Linnet, Reed Bunting, Tree Sparrow and Dunnock (Figure 9.3, Table 9.16). Bird numbers, especially in ELS and ELS-X transects, could be highly variable between different transects within the same Treatment generating large standard deviations (Table 9.16, and Table 9.13). This is considered further in the discussion. In the baseline winter of 2005/06 there were significant Treatment effects for 'granivorous species', Chaffinch and Great Tit, with greatest numbers occurring in ELS and least in ELS-X. In 2006/07, the trend for larger numbers of birds in ELS and ELS-X was significant for Reed Bunting, Yellowhammer and Tree Sparrow and marginally so for 'granivorous species' and Dunnock. In 2009/10 and 2010/11, there were significantly more birds in the groups of 'all species' and 'granivorous species' in ELS-X, and this was largely due to significantly greater numbers of Chaffinches and Linnets and to a lesser extent, Reed Buntings and Dunnocks (Table 9.16). Other significant Treatment effects in 2009/10 were for Blue Tit (smallest numbers in ELS), and in 2010, for Wren (most birds in ELS-X) and Blackbird (least birds in ELS).

Table 9.16 Within-year Treatment effects on numbers of birds per kilometre of transect in winter. Effects significant up to approximately the 10% level, and non-significance where appropriate, are shown (all other results not significant).

			Numbers of birds, mean ± SD		
Species/group	$F_{12,14}$	P	CC	ELS	ELS-X
All species					
2009/10	4.59	0.033	57.5 ± 12.9	54.2 ± 23.1	303.0 ± 366.0
2010/11	15.00	< 0.001	62.7 ± 40.9	59.2 ± 25.8	208.7 ± 69.1
Granivorous species					
2005/06	3.72	0.055	11.4 ± 4.2	13.0 ± 5.6	5.4 ± 3.3
2006/07	2.87	0.096	26.2 ± 8.02	58.7 ± 29.9	68.1 ± 41.8
2009/10	5.63	0.019	13.8 ± 9.3	19.6 ± 15.6	259.6 ± 365.0
2010/11	13.33	< 0.001	23.6 ± 35.1	26.2 ± 19.7	161.9 ± 65.7
Chaffinch					
2005/06	4.57	0.033	9.7 ± 3.4	10.8 ± 4.4	4.7 ± 2.5
2006/07	-	NS	-	-	-
2009/10	5.32	0.022	10.0 ± 3.3	13.0 ± 11.7	100.4 ± 122.6
2010/11	6.61	0.012	22.5 ± 35.5	12.7 ± 6.4	87.4 ± 48.3
Linnet					
2005/06	Insufficient data (3 birds in 2 transects)				
2006/07	-	NS	-	-	-
2009/10	6.54	0.012	0	3.4 ± 6.4	157.0 ± 241.9
2010/11	4.03	0.046	0	6.8 ± 14.0	58.8 ± 91.8
Reed Bunting					
2005/06	No birds recorded in this winter				
2006/07	4.69	0.031	0.35 ± 0.79	3.4 ± 2.5	6.3 ± 7.0
2009/10	Insufficient data (4 birds in 3 transect)				
2010/11	9.21	0.004	0	1.6 ± 9.9	9.9 ± 13.3
Yellowhammer					
2006/07	3.18	0.078	4.7 ± 4.6	23.6 ± 23.9	7.0 ± 3.7
All other winters = NS					
Tree Sparrow					
2006/07	14.49	< 0.001	0	0	2.1 ± 2.6
All other winters = insufficient data (between 1 and 3 birds in 1 or 2 transects)					
Dunnock					
2005/06	-	NS	-	-	-
2006/07	2.62	0.114	9.2 ± 1.8	11.5 ± 7.1	18.4 ± 5.4
2009/10		NS			
2010/11	8.36	0.005	2.3 ± 2.0	5.1 ± 1.6	8.2 ± 3.3
Blackbird					
2010/11	3.42	0.067	10.5 ± 1.2	5.4 ± 3.3	10.9 ± 5.5
All other winters = NS					
Wren					
2010/11	3.75	0.054	1.6 ± 1.7	1.7 ± 0.42	2.8 ± 0.43
All other winters = NS					
Great Tit					
2005/06	3.62	0.059	2.4 ± 2.2	4.9 ± 2.9	0.67 ± 1.5
All other winters = NS					
Blue Tit					
2009/10	3.24	0.075	12.7 ± 5.9	7.0 ± 2.3	10.6 ± 2.3

9.4.4 Bird Use of Winter Food Patches

The abundance and species richness of birds counted on the bird food patches were substantially higher than those recorded in equivalent areas of crop. In 2007/08 (Biennial mixture in its second year), the mean number of birds and the mean species richness per patch per visit (all patches combined) standardised to an area of 1 ha was 79 and 8.7 respectively compared with 0.3 birds and 0.2 species in 1 ha of crop. The species recorded in the crop were Skylark *Alauda arvensis* and Grey Partridge *Perdix perdix*.

The numbers of birds counted varied between patch types (Appendix 12.4, Figure 2), but due to large variation in the numbers present on individual patches (see below), the only significant difference occurred in November ($F_{2,16}$ = 8.587, P = 0.003). Overall, the Bumblebird mixture attracted fewer birds and in general, bird numbers on all patch types declined in late winter (late January, February). There was large variation in the responses of individual species to the different patch types; for examples see Figure 3 in Appendix 12.4. Apart from an initial presence in Deluxe patches in early winter, Yellowhammers were rare whereas Linnet numbers were maintained in Deluxe until late winter (Appendix 12.4, Figure 3a and 3b). Song Thrushes showed an increasing trend in the Biennial mix until late winter (Appendix 12.4, Figure 3c) and Dunnocks occurred in all three patch types (Appendix 12.4, Figure 3d). Chaffinches (Appendix 12.4, Figure 3e) showed a similar pattern to that for all species combined (Appendix 12.4, Figure 2) while Greenfinches (Appendix 12.4, Figure 3f) showed a similar response to Yellowhammers to Deluxe and a peak in numbers in the other two types in December. Goldfinches (Appendix 12.4, Figure 3g) were commonest in the Biennial mix, declining in numbers here throughout the winter, but were, at least in part, attracted by chicory *Cichorium intybus* which had occurred as an accidental contaminant. Reed Buntings (Appendix 12.4, Figure 3h) occurred sporadically in low numbers, but were absent from the Biennial mix apart from a single bird in one patch in early January. As with the results for all species combined, the large variation in counts of individual species between individual patches caused most differences between patch types to be either non-significant or to occur sporadically in different months for different species (Appendix 12.4, Table 3). The exceptions were the increasing and/or decreasing trends for Song Thrush, Chaffinch and Goldfinch (Appendix 12.4, Figures 3c, e and g).

A positive relationship between the amount of seed produced by a patch and the total numbers of birds present at the beginning of the winter was expected. However, the relationship was significant in November only (slope = 0.00152, $F_{1,18}$ = 8.297, P = 0.010, Appendix 12.4, Figure 4), the best model using a single slope and intercept. There was no evidence of a relationship in either October (slope = -0.0003, $F_{1,16}$ = 0.095, P = 0.762) or December (slope = 0.00127, $F_{1,16}$ = 1.212, P = 0.288). This was probably due to the large variation in seed yield between patches, including within the same type, and also between bird counts on individual patches (Appendix 12.5, Figs. 3, 4 and 5). Relationships between seed yields and bird counts later in the winter were not expected (due to the length of time from measurement of the seed crop in September) and none were found.

9.4.5 Bird Use of Other Prescribed Habitats in Late Winter

The largest numbers of birds were recorded on the bird food patches (Appendix 12.4, Table 2) with only small numbers recorded on any of the other margin or patch habitat types. The most frequent species on tussocky margins were Goldfinch and Reed Bunting, the former being attracted to teasels *Dipsacus fullonum*. In pollen and nectar margins, partridges (mostly Red-legged Partridge *Alectoris rufa*) and Chaffinch were most frequent. No species were associated with any of the other habitat types, and in all cases, the numbers of birds present was small (Appendix 12.4, Table 3).

9.4.6 Tit Breeding Success

Over the four years, 2007 to 2010, there was a total of 272 initial breeding attempts (170 Great Tit, 102 Blue Tit), an attempt being defined as activity progressing as far as the laying of at least one egg (and excluding replacement attempts and second broods). Of nests that progressed at least as far as a complete clutch, the ratio of Great Tit to Blue Tit box occupation was: 2007 = 33:31; 2008 = 46:24; 2009 = 42:26; 2010 = 45:21. The greater proportion of Blue Tits in 2007 was not significant across all years (χ^2_3 = 0.747, P = 0.863), but in terms of pairs of years, did differ from 2010 (χ^2_1 = 3.894, P = 0.048).

Temperature and rainfall differed substantially between the four years (Figure 9.4) and this had consequences for the birds' breeding success. Both 2007 and 2008 had high average spring rainfall (March-May), and particularly so in 2007 in May, coinciding with the main period of nestling growth. Both 2009 and 2010 were comparatively drier over the same period. Spring temperatures were highest in 2007 and 2009, with 2008 having a

warmer May, and 2010 being cooler on average (Figure 9.4). One-way ANOVAs showed that first egg date was significantly earlier in 2007 and 2009 than in the other two years, for both species (Table 5).

Breeding parameter	Habitat	2007	2008	2009	2010
GREAT TIT					
1st egg date	Farmland	20.7 (33)	30.2 (46)	20.5 (42)	26.8 (45)
	Woodland	17.1 (55)	27.3 (44)	15.1 (38)	22.0 (45)
Clutch size	Farmland	8.0 (33)	7.5 (47)	8.8 (39)	8.7 (41)
	Woodland	9.4 (56)	8.9 (42)	10.2 (38)	9.4 (42)
Mean nestling body mass, g	Farmland	14.8 (22)	13.6 (31)	15.1 (34)	15.6 (34)
	Woodland	17.8 (51)	17.0 (33)	17.3 (36)	16.7 (36)
Number fledged	Farmland	5.1 (16)	4.1 (28)	6.1 (33)	6.6 (31)
	Woodland	8.7 (50)	7.6 (33)	9.7 (36)	8.5 (35)
% success	Farmland	61.8 (16)	52.9 (28)	70.9 (32)	77.7 (31)
	Woodland	94.0 (50)	87.3 (33)	94.3 (35)	91.6 (35)
BLUE TIT					
1st egg date	Farmland	20.9 (31)	29.3 (24)	19.0 (26)	25.1 (21)
	Woodland	18.9 (17)	26.2 (26)	15.0 (19)	20.7(22)
Clutch size	Farmland	9.3 (31)	10.2 (24)	10.9 (25)	10.8 (20)
	Woodland	10.0 (17)	10.0 (24)	12.1 (18)	11.0 (22)
Mean nestling body mass, g	Farmland	9.4 (28)	8.8 (19)	9.5 (22)	9.9 (17)
	Woodland	10.6 (17)	9.9 (22)	10.7 (16)	10.3 (20)
Number fledged	Farmland	5.0 (22)	4.7 (18)	8.1 (21)	7.7 (17)
	Woodland	8.4 (17)	8.0 (22)	11.2 (16)	9.9 (20)
% success	Farmland	51.8 (22)	45.5 (18)	74.9 (21)	71.5 (17)
	Woodland	83.2 (17)	78.8 (22)	91.3 (16)	88.5 (20)

Table 9.17 Comparison of tit breeding success on farmland and in woodland. Data are for successful nests only at each breeding stage. Number of nests given in parenthesis; for 1st egg date, April 1st = 1.

All other breeding variables, except the number of young alive in the nest at 11 days of age for Great Tits, also showed significant differences across years (Table 9.17). Tukey *post hoc* tests showed that in most cases these differences were attributable to a separation between 'poorer' breeding performance in 2007 and 2008 *versus* 'better' performance in 2009 and 2010. Within-year differences between Treatments were uncommon and showed no particular pattern. In 2007, clutch size in Great Tits was greater in ELS than in either CC or ELS-X (Mean ± SD: CC = 7.4 ± 1.4; ELS = 9.1 ± 1.2; ELS-X = 7.8 ± 1.2). In 2010, first egg date in Great Tits was earlier in ELS than either CC or ELS-X (April 1st = 1: CC = 24.8 ± 6.5; ELS = 29.5

± 3.4; ELS-X = 25.5 ± 6.27). Also in 2010, clutch size in Blue Tits was smaller in CC than either ELS or ELS-X (CC = 9.9 ± 1.8; ELS = 11.0 ± 1.0; ELS-X = 11.5 ± .9).

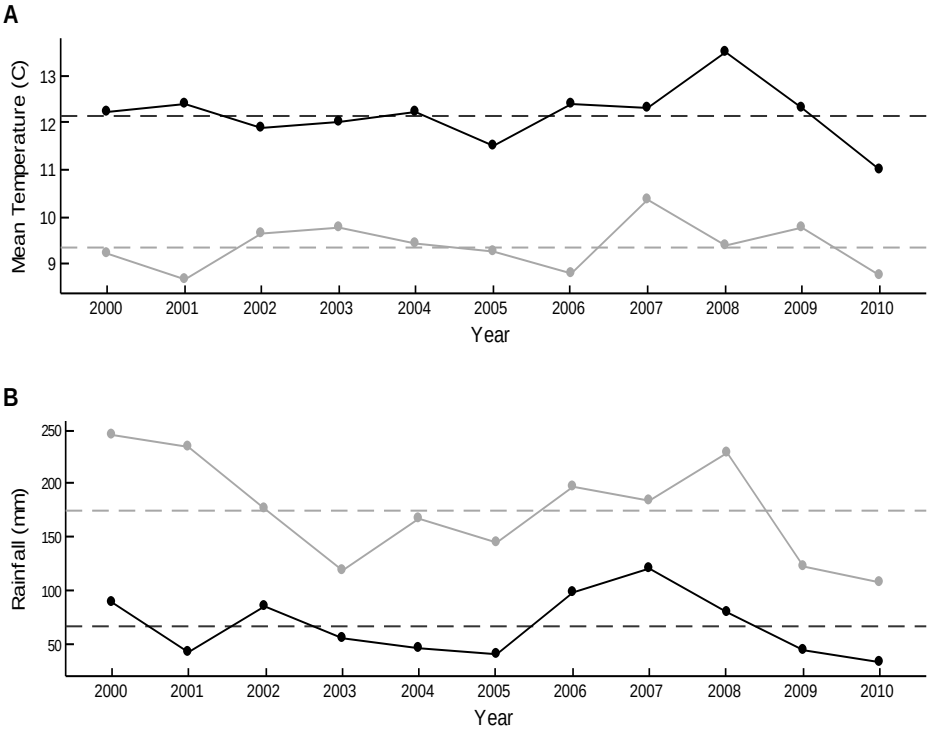


Figure 9.4 Mean monthly (**A**) temperature, ^{o}C, and (**B**) rainfall, mm, for 2000 to 2010. For both graphs, solid, black lines and points indicate values for May; solid, grey lines and points indicate values for the spring months of March, April and May combined. Dashed lines in corresponding colours indicate the mean values across all 11 years.

Despite significant inter-annual variation, a number of habitat variables influenced most of the breeding parameters for both species, the exceptions being the number of young alive in the nest at 11 days of age for Great Tits (also no year effect) and first egg date for Blue Tits (Table 9.18). Most best models used the 100 m radius buffer around the nest box and habitat variables associated with the tree canopy, usually tree proximity index and tree canopy volume, were selected most frequently. The two species showed some different responses; for Great Tits, tree canopy variables were associated with all breeding parameters whereas proximity to ES margin/patch habitat was significant for Blue Tits for the number of young fledged and overall success (Table 9.18). Timing of breeding, i.e. first egg date was affected by habitat in addition to year in Great Tits, but not for Blue Tits where only year was significant. Clutch size in both species was affected by habitat, but the best model for Great Tits was for a 50 m radius around the nest box rather than 100 m as in Blue Tits. Average weight and biomass in both species were influenced by similar habitat variables, all associated with the tree canopy. The number of fledglings and overall success

in Great Tits were again mostly associated with the tree canopy within 100 m of the nest box, whereas the best predictors for Blue Tits were associated with ES margin/patch habitat within 50 m. The best models for each of the four years separately were in broad agreement with those reported here for the whole dataset and are not shown.

As expected, both species on farmland bred later, laid smaller clutches (with the single, and non-significant, exception of Blue Tits in 2008) and fledged fewer and lighter young than was the case for woodland breeding tits in all four years (Table 9.17).

Breeding parameter	Explanatory variables	Radius, m	n	Adj. r^2	P
GREAT TIT					
1st egg date	Year Tree canopy proximity index Hedge mean height	100	163	40.8	< 0.001
Clutch size	Year Tree canopy volume Hedge volume (-ve) Margin/patch proximity index	50	156	16.8	< 0.001
Nestling nos. at 11 days	(Year, NS) (Hedge volume, NS)	100	120	2.9	NS
Mean nestling body mass at 11 days, g	Year Tree canopy volume Tree maximum height	100	119	20.7	< 0.001
Total biomass at 11 days, g	Year Tree canopy proximity index Margin/patch proximity index (-ve)	100	119	11.1	0.011
Number fledged	Year Tree canopy proximity index	100	106	21.6	< 0.001
% success	Year Tree canopy proximity index Hedge volume Tree mean height	100	105	23.9	< 0.001
BLUE TIT					
1st egg date	Year	100	97	44.3	< 0.001
Clutch size Tree canopy proximity index Habitat diversity (S-W) index	Year	100	96	21.8	< 0.001
Nestling nos. at 11 days	Year Tree maximum height	100	83	16.8	< 0.001
Mean nestling body mass at 11 days, g	Year Tree canopy proximity index Tree canopy volume (-ve)	50	83	17.6	< 0.001
Total biomass at 11 days, g	Year Tree canopy proximity index	100	83	21.4	< 0.001
Number fledged	Year Margin/patch proximity index	50	76	30.7	< 0.001
% success Margin/patch proximity index	Year	50	76	26.5	< 0.001

Table 9.18**.** Summary of best-subsets regression models showing significant variables influencing tit breeding performance for all four years combined. All relationships are positive except where indicated, i.e. –ve = negative. S-W index = Shannon-Weiner habitat diversity index.

9.5 **Discussion**

9.5.1 Breeding and Winter Birds

For most individual species and species groups, the main change in the numbers of breeding territories across the four years was an increase in 2007, the first breeding season following establishment of the management prescriptions (Table 9.3). That this increase was not maintained for most species was probably due, at least in part, to the weather in subsequent winters, and especially that of 2009/10. The winter of 2007/08 (http://www.metoffice.gov.uk/climate/uk/2008/winter.html) was characterised by average or above average temperatures (compared to the 1971 – 2000 average) whereas that for 2008/09 had below average temperatures in December and January (http://www.metoffice.gov.uk/climate/uk/2009/winter.html). In England, it was the coldest winter since 1995/96. The following winter of 2009/10 was even colder, with temperatures well below average in December, January and February making it the coldest winter in England since 1978/79 (http://www.metoffice.gov.uk/climate/uk/2010/winter.html). The response of Whitethroat, a summer migrant breeder and hence unaffected by British winter weather, was consistent with this hypothesis in that it showed no particular overall trends and lacked the increase in 2007. The numbers of Lesser Whitethroat, another migrant, were too small to draw conclusions.

The numbers of birds present in the winter also increased following the establishment of the management prescriptions, but unlike the breeding birds, numbers remained elevated for the rest of the study for several species, most especially Chaffinch and Linnet (Table 9.13). The winter data included an additional winter, i.e. 2010/2011, which was again severe (http://www.metoffice.gov.uk/climate/uk/2011/winter.html). Although less cold on average across the whole winter than 2009/10, it was 5.1 ^{o}C colder than average in December 2010 compared to 2.0 ^{o}C below average in December 2009, making it the second coldest winter (after 2009/10) in England since 1995/96. This run of colder than average winters had different effects on different species. Chaffinch and Linnet numbers increased, probably as a consequence of their use of the bird food patches; both these species were the most consistent and abundant species recorded on the patches (Appendix 12.4, Fig. 3). Also, where there were significant Treatment effects on inter-winter differences, greater numbers occurred most frequently in ELS-X for Chaffinches and

'granivorous species' (Table 9.14 and Table 9.14). That Chaffinch numbers in winter increased without a concomitant increase in the breeding population (except in 2007) suggested that the availability of winter seed supplies was attracting birds from a wider area during these cold conditions. The more sporadic response of Reed Bunting and Yellowhammer suggested that birds might be moving on a larger scale in response to both the type and quantity of seed available, both at Hillesden and elsewhere (Butler *et al.* 2010, Siriwardena 2010). Such shifting movements are typical of finches and buntings in winter (Prŷs-Jones 2002). The increased numbers of tits in the later winters (significant for both species), compared to more stable breeding numbers, also suggested a winter influx which seemed strange in that they were seldom recorded in the bird food patches or other managed habitats (at least in the late winter of 2008) and farmland in winter is more exposed to severe weather than woodland. Blue Tits are known to respond to the availability of winter roost sites (Dhondt *et al.* 1991) and thus the presence of the nest boxes may have had an influence. Artificial foods such as peanuts supplied by householders may also have affected tit abundance and movements in the landscape. For several other species, including Song Thrush, Wren and Robin, the decline in numbers in the colder winters suggested direct mortality, as did the decline in Dunnock numbers after the increase in 2006/07. This winter mortality might also have had a knock on effect, reducing breeding numbers of these species, especially for Robin.

For the breeding birds at Hillesden, between 2006 and 2007, there were increasing trends for six of the individual resident species, little change for seven and no declines (Table 9.3, and Table 9.19). This compared with increasing trends for two species, little change for four and declines for seven for national population trends in England over the same period (Table 9.19, Riseley *et al.* 2008). Changes in national populations from 2008 to 2009 (Riseley *et al*. 2010) showed declines for eight species, six of which were significant and again indicated an effect of winter weather. At Hillesden, the breeding census was not carried out in 2008 making trends to 2009 difficult to interpret compared to the national data (Riseley *et al.* 2009), but there was overall agreement with the declining trends except for Linnet, Blackbird, Song Thrush and Great Tit which maintained or increased in numbers at Hillesden (Table 9.19). Between 2009 and 2010 at Hillesden, there were increasing trends for one species, little change for seven and declines for five. National trends appeared to

have fared better with seven increasing trends, two with little change and two declines (Risely *et al.* 2011). The largest decline at both Hillesden and nationally was for Robin. National populations appeared to making something of a recovery after general declines in the previous two years, whereas populations at Hillesden did not. This difference may have been related to regional (and habitat) differences in the severity of the winter, but given the different scales of the two studies comparisons can only be of a general nature. Linnet was the notable exception to the general trend for breeding birds at Hillesden to peak in 2007, the increase in numbers of territories in 2007 (more than doubled) being sustained and increased through to 2010. In all years subsequent to 2006, the increase between years was due to more territories in ELS, and also in ELS-X in 2009, in contrast to no change in CC, indicating a positive effect of the former two Treatments. Given this positive response of Linnet to the management prescriptions, a greater effect of ELS-X compared to ELS might have been expected. However, the observed results may have been due to the scale of the experiment relative to bird territory sizes and foraging distances and for the tendency of this species to nest in loose groups. The transect-level habitat relationships suggested that Linnet territories were more abundant in hedges with fewer trees, at least in some years (Table 9.6) which could have influenced nest site selection and contributed to scale effects. There may also have been a crop effect in relation to the availability of oil seed rape as a food source (Moorcroft & Wilson 2000) – this will be investigated further in additional analyses. In contrast to the positive response of Linnet, Reed Bunting showed no appreciable change across the five years suggesting that this species might have been limited by factors other than those manipulated by the experiment and/or was responding at a different scale. Of the other seed eaters, the increases in Chaffinch and Yellowhammer in 2007 suggested that, under certain environmental conditions at least, the management options could have a positive effect at a whole-farm scale. Nationally, both these species showed a declining trend between 2006 and 2007, as did Linnet while Reed Bunting showed little change (Table 9.19).

Table 9.19 Breeding season bird abundance (numbers of territories per kilometre of transect, n = 15) compared with the % change in national bird populations in England between 2006/07, 2007/08 and 2009/10. National data are from the BTO BBS (Risely *et al*. 2008, 2009, 2010, 2011) and show changes between adjacent years. Significant change indicated by *. Hillesden data for 2008 are included but must be treated with caution due to the different methodology used to estimate territories during the collection of productivity data – see text.

Species/group	2006	2006 – 07 England (%)	2007	2007 – 08 England (%)	2008	2008 – 09 England (%)	2009	2009 - 10 England (%)	2010
Chaffinch	7.3	-3	9.1	-5	6.7	-1	6.5	+3*	6.6
Linnet	0.7	-13	1.6	+19*	1.7	-4	2.3	+4*	1.8
Reed Bunting	1.2	+1	1.1	-3	1.8	+10	1.2	+2	0.9
Yellowhammer	3.8	-9	4.1	-3	3.5	+1	3.0	+2	3.2
Blackbird	3.0	+4	3.2	+5	4.4	-5*	4.5	0	3.9
Song Thrush	0.6	+1	0.7	+6	1.5	-5*	1.1	-4	1.0
Great Tit	2.3	-5	2.6	-1	-	-7*	2.5	+12*	2.4
Blue Tit	2.8	-7	3.6	-1	-	-5*	2.3	+20*	2.3
Dunnock	3.7	+1	4.6	-6	3.6	+1	3.0	+9*	2.5
Robin	2.8	+2	3.5	+8*	3.0	-4*	1.8	-9*	0.8
Wren	2.1	+12	2.2	-1	4.0	-10*	1.8	-5*	1.7
Whitethroat	3.1	-7	2.8	+1	3.6	+7*	3.1	+10*	3.8
Lesser Whitethroat	0.5	-3	0.7	-1	-	-2	0.4	+9	0.2

After non-significant increases in 2007, Dunnock, and especially Robin, declined significantly while Wren also showed a small declining trend after 2007. As discussed above, these changes were most likely due to the winter weather. The lesser decline in Dunnocks compared to Robins might have been related to the former species use of the bird food patches, especially the greater cover and potential food supplies, offered by the Biennial patches (Appendix 12.4, Fig. 3d). Numbers of breeding tits appeared to be maintained, after an increase in Blue Tit in 2007. Reproductive success of tits on farmland is poor compared to woodland (see below) and thus the habitat may comprise a population sink with numbers replenished by immigration (perhaps as illustrated by the numbers present in winter). National populations of both Great and Blue Tit have declined year on year from 2006/07 to 2008/09 (Table 9.19, Riseley *et al.* 2008, 2009, 2010), but long-terms trends are still generally increasing; both species are amongst the most numerous in the UK and easily able to supply a source of immigrants.

Clear-cut treatment effects on inter-year changes in numbers were relatively infrequent, the clearest being for breeding Linnets and for Chaffinches and 'granivorous species' in winter. These results suggested that the presence of the bird food patches in ELS and ELS-X had had a positive effect on the increase in the numbers of Linnet territories and the increases in the numbers of seed eaters present in the winters after the baseline in 2005/06. Significant within-year Treatment effects were sparse in the breeding season, but the greater numbers of territories of seed eaters in ELS-X (and ELS) in the later years of 2009 and 2010 were consistent with a positive effect of the bird food patches (as were the non-significant trends for several other groups and species, Figure 9.1). The strength of such effects tended to be greatest in 2009 and 2010 which suggested a time-lag in the response of the birds to the provision of resources and/or in the detectability of the response (Siriwardena 2010). Detecting within-year Treatment effects in the winter was more problematic due to greater inherent variation in the data. The birds, especially the seed eaters, had clumped distributions concentrated, as expected, around the bird food patches, generating the chance of large numbers should a flock be present during a census or possibly few birds if a flock had naturally moved on or been disturbed. Several other species, especially tits, were also more inclined to occur in flocks in the winter, again creating the potential for large variation. Despite these difficulties, where Treatment effects occurred, there was a general tendency for greater numbers, especially of seed eaters, to be

present in ELS-X and ELS in the later years of the study. The presence of a reliable food source over several years may have influenced bird behaviour across years, but patterns of usage and abundance may also have been modified by the colder winters of the later years.

Relationships between numbers of territories and habitat availability at both transect and wider landscape-scales were generally as expected from species ecologies, i.e. species connected with woodland and woodland edge habitats such as Blue Tit and Robin tended to have positive relationships with trees, hedges and woody vegetation whereas those more typical of open farmland such as Yellowhammer and Linnet had positive relationships with margins and patches and arable land and negative ones with trees and woody vegetation. Relationships between seed eaters and the availability of margins and patches were most frequent in the last two years of the study suggesting, as above, a time-lag in the birds' responses. The largest number of positive relationships between the seed eaters and margin and patch habitat occurred in 2009, suggesting a build-up in response across the years, with the subsequent reduction in 2010 perhaps reflecting the run of cold winters. It is also possible that the birds' responses, in terms of both numbers and locations of territories, could be influenced by patch quality in a given season and resource availability in the wider landscape. These two factors might be particularly important in severe winters in relation to short and mid-distance movements in winter and settling decisions in spring. The positive responses of both Blackbirds, Song Thrushes and Whitethroats to margins and patches in 2009 suggested that these habitats offered foraging opportunities to more than just seed eaters.

The responses of some species such as Linnet, Dunnock and Whitethroat suggested a preference for hedges coupled with an avoidance of trees, the category of 'all woody vegetation' at both scales being dominated by the area of tree canopy. At the wider landscape-scale, the directions of responses to 'arable land' and 'agricultural grassland' (especially negative relationships with grassland) were probably driven by the strong correlation between these two land-use types (Table 9.8).

9.5.2 Breeding Bird Productivity

The greater productivity in 2010 than in 2008 was similar to national trends and probably reflected the consequences of the wetter summer in 2008 (141% of the 1971 to 2000 average, compared with 107% in 2010, http://www.metoffice.gov.uk/climate/uk/). The

distribution of the rainfall in 2008 (wetter than average in all three months of June, July and August versus most rain later in the summer in 2010) and the drier spring in 2010 (usually favourable for tits and other relatively early breeders) were also likely to have favoured 2010. Although productivity was significantly greater in 2010 for all three species groups and for several individual species at the whole farm level (Table 9.10), there were no Treatment effects on these increases between years.

Similarly, there were few within-year Treatment effects on any of the measures of productivity. The only result significant at less than P = 0.05 was for the numbers of broods per km for Blackbird in 2008 in which greater numbers occurred in CC. This measure of Blackbird productivity was also positively related to the area of buildings within 100 m of the transects (Table 9.12). The area of buildings acted as a proxy for the presence of gardens suggesting that such habitat might offer foraging opportunities for breeding Blackbirds. The 100 m buffer zone for one of the CC transects included a larger than average proportion of gardens and generated positive relationships for several species for both numbers of territories and the measures of productivity, but the significance of these relationships was due entirely to this outlier effect (except for Blackbird and Dunnock as indicated in Table 9.12). The lack of Treatment effects could have been affected by scale, and wider landscape effects, if foraging distances frequently exceeded the scale of individual Treatments. The apparent lack of response of productivity to the availability of margin and patch habitat at both the transect and wider landscape-levels also suggested conflicts with scale. Similarly, while relationships for seed eaters with 'arable land' were positive, as might be expected for farmland birds, the few relationships with margin and patch habitat that were apparent were negative (Table 9.11 and Table 9.12). In the assessment of tit breeding success (see below), habitat analysis based on a distance of 100 m around the nest site appeared to be more appropriate than 50 m, but a 100 m buffer at the whole transect level may be inappropriate given the length of the transects. It should also be noted that for numbers of territories, relationships with margin and patch habitat were generally positive for seed eaters at both transect and wider landscape-levels (Table 9.6 and Table 9.7), suggesting a better match between response and predictor variables at this scale. Thus, a more detailed analysis using habitat availability at the level of individual territories could be more informative for the productivity measures, but is beyond the scope of this report.

It is also possible that the method was not sensitive enough to accurately record productivity, i.e. that broods were missed, and/or that brood size (in addition to or instead of numbers of broods) varied with Treatment. There may also be complex relationships between breeding effort (numbers of failed nesting attempts), predation and productivity which dilute direct habitat effects. Numbers recorded in the transects could also have been influenced by birds moving in from adjacent habitat. Differences between Treatments (and territories) might occur in terms of the quality (e.g. body mass, degree of development at fledging) of young produced rather than the absolute numbers; in general, heavier young are those most likely to survive (e.g. Tinbergen & Boerlijst 1990, Donald 2004). Despite these potential difficulties, the remote sensed data should facilitate future analysis of habitat characteristics associated with success or failure at the level of individual territories and the availability of likely foraging habitat.

9.5.3 Winter Food Patches and Other Prescribed Habitats in Late Winter

As expected, many more birds were counted on the bird food patches than on equivalent areas of crop (e.g. Boatman *et al.* 2000, Henderson *et al.* 2004, Stoate *et al.* 2004, Field *et al.* 2009), but the winter census based on the 15 transects, and in 2006/07 in particular, indicated an increase in bird numbers across the farm in general, not just on the patches. This implied that birds do not simply move between food-rich patches across the landscape, but also use intervening habitat of different types. A response to habitat other than, or in addition to, the bird food patches was also indicated by the results for Yellowhammer (Whittingham *et al.* 2005) and Blue Tit. Both these species showed a significant increase in numbers in the winter of 2006/07 (Appendix 12.4, Table 4), but neither was present in large numbers on the bird food patches (App. 11.4, Figure 3). Overall, the species showing the greatest response to the patches in winter were, as expected, the seed-eaters, plus Song Thrush and Dunnock (see below). With the exception of Blue Tit (which appears to be less sensitive to habitat quality than Great Tit, Hinsley *et al.* 1999), woodland or woodland edge species such as Robin and Wren tended to show little response.

Bird exploitation of dedicated food patches will depend on many factors including seed type and yield, patch location, and the distribution and abundance of alternative food supplies in the wider landscape (Whittingham & Evans 2004, Siriwardena & Stevens 2004, Stoate *et al.* 2004, Siriwardena *et al.* 2006, Siriwardena 2010). Some species, such as

Dunnock and Song Thrush, may also respond to attributes other than those directly related to the sown seed crop such as shelter, access to damp ground and associated weed seed and invertebrate food resources (Peach *et al.* 2004, Gilroy *et al.* 2008). This was thought especially pertinent for Song Thrush (and other thrushes) in the tall dense cover offered by the Biennial patches (Appendix 1, Figure 3, Table 3). Similarly, the presence of Linnets in the Deluxe patches (Appendix 1, Figure 3b) was thought to be due in part to growth of weeds (e.g. chickweed *Stellaria media* and groundsel *Senecio vulgaris*) and, as mentioned above, Goldfinches responded to the accidental presence of chicory in the Biennial mix (Appendix 1, Fig. 3g, Table 3). The range of responses of individual species to the different seed mixtures used at Hillesden indicated that selecting a single 'best' mix was unlikely to be feasible; different bird species are well known to prefer different seed types and sizes (Diaz 1990, Wilson *et al.* 1999, Boatman & Stoate 2002, Stoate *et al.* 2004, Holland *et al.* 2006, Perkins *et al.* 2007). There is also the choice of annual versus biennial mixes; the results for the Biennial mix reported here were dominated by the characteristics of the second year-seeding species, kale and fodder beet. Therefore, provision of a range of patch types, within a single farm or co-ordinated across larger areas, could be most cost-effective.

The amount of food resource available in a patch is also a major determinant of overall bird use and thus patch quality in terms of coverage and yield is of major importance. Variation in seed production between individual patches (Appendix 12.5, Figs 3 and 4) was substantial. For example, the least productive Biennial patch produced about 90% less seed (for sown species) than the most productive one. For each patch type, if all patches had performed as well as the most productive, the total yield of sown species could have been increased by about 64%. The large variation in both seed yields and bird counts between patches obscured the expected positive relationship between yield and bird numbers except in November 2007 (Appendix 12.4, Fig. 4). However, using this relationship and the difference between actual mean seed yield per patch and the estimated "best" performance (i.e. an increase of 64%), if all patches had achieved "best" performance then such a yield would predict an increase in bird numbers of 48%. Seed production is also complicated by the presence of unsown species, some of which such as chickweed and groundsel are used by the birds, but overall there was a negative correlation ($P = 0.001$) between the yields of sown and unsown species (see Appendix 12.5). Thus the quality of ES

options is likely to be as important as their identity. Bird food patches, and other non-crop habitat types, tend to be located in the least favourable/least productive areas of fields, and this, coupled with the repeat sowing of the same areas across several years can result in poor establishment and performance. Therefore, the production and maintenance of good quality bird food, and other semi-natural habitat, will probably require a similar management effort to that devoted to crops (Stoate *et al.* 2004, Siriwardena & Anderson 2007, Douglas *et al.* 2009, Lobley *et al.* 2009).

Despite the large variation in the use of the three seed mix types (Appendix 12.4, Figures 2, 3 and 4), they all showed a sharp drop in bird numbers in late January and early February, the beginning of the so-called 'hungry gap' (Siriwardena *et al.* 2008) when both managed and natural food supplies become depleted and/or less accessible to some species due to seed drop. Seed depletion in the Hillesden patches varied to some extent between species (most rapid loss in kale and millet, and slowest in fodder beet), but was generally rapid from early to mid winter (Appendix 12.5, Figures 1 and 2). By January 1st, mean depletion across all seed mixes was about 80% and exceeded 90% by mid January. Filling this gap may require novel seed crops or greater areas of dedicated bird food and there is also the possibility of 'artificial' feeding. However, the latter often involves the use of cereals and such relatively large and hard seed may not be suitable for all species. Even though sown crops may be depleted in late winter, providing these additional/alternative resources earlier in the winter may prolong the availability of 'natural', i.e. non-cropland, food supplies. The other types of ES habitat patches and margins were little used by birds in late winter, but this was perhaps not surprising given that they are not designed to provide winter foraging habitat for birds. It is possible that these habitats are used by birds at other times in the winter or late autumn (as well as in spring and summer) and additionally also contribute to conserving alternative food supplies. It might also be possible to increase their value as habitat for birds in winter by, for example, allowing grass margins to go to seed (Buckingham & Peach 2006) and including additional plant species such as Teasel. However, the concentration of birds on the dedicated seed patches does strengthen the case for promoting winter bird food patches as a preferred ES option. There is also the question of where the birds go in late winter. Our observations suggest that they move more widely in the landscape than the farm-scale and this is consistent with various studies of winter food

use (Robinson *et al.* 2004, Siriwardena *et al.* 2006) and of winter bird movements in general (e.g. Prŷs-Jones 2002). This in turn suggests that consideration of local landscape factors and co-ordination of ES management at greater than a farm-scale will be necessary to maximise effectiveness for farmland birds (Siriwardena 2010).

9.5.4 Tit Breeding Success

As found in other studies of tits breeding in secondary habitats (e.g. Cowie & Hinsley 1987, Riddington & Gosler 1995, Hinsley *et al.* 1999, 2008 & 2009), reproductive performance of both Great Tits and Blue Tits breeding in the Hillesden farmland hedgerows was poorer than that of woodland breeders (Table 9.17). This was true in terms of both the numbers and quality of young fledged. In general in tits, the young most likely to survive to enter the breeding population are those which fledge early and with the greatest body mass (Tinbergen & Boerlijst 1990, Verhulst *et al.* 1995). The Hillesden fledglings, especially Great Tits, were notable lighter in weight. Overall success was also much reduced compared to woodland and this was influenced in large part by the weather. Birds on farmland are much more exposed to adverse weather conditions increasing the risk of total brood failure, for example, as occurred in 2007. During 2-3 days of heavy rain and high winds over the late May bank holiday weekend in 2007 44% of the young in the nest died. Of 354 chicks alive at 11 days of age, only 198 were alive at 20 days. The poor weather occurred when many of the broods were nearly ready to fledge and thus at an age when their food demand was greatest, making them vulnerable to poor parental foraging conditions. Such effects were also evident in woodland populations in the same year, but were less extreme, due to a greater degree of shelter in woodland and probably a better food supply. The overall better performance in 2009 and 2010 was also probably a consequence of drier conditions during chick rearing in these two springs (Figure 9.4).

Although breeding started a little later on farmland than in woodland, the pattern of earlier or later breeding across the four years was similar in both habitats. Tits attempt to time their breeding to maximise access to tree-dwelling caterpillar larvae which comprise the bulk of the nestling diet (at least in woodland) and thus breed earlier in warmer springs when both vegetation and caterpillar development are also earlier (Perrins 1970, McCleery & Perrins 1998, Hinsley *et al.* 2006). This can be seen in the Hillesden data where breeding

started significantly earlier in the two warmer years of 2007 and 2009 (Table 9.17, Figure 9.4).

The influence of weather conditions on tit breeding success was also indicated by the models attempting to relate breeding performance to habitat structure around the nest (Table 9.16) where year was significant for all models bar that for the number of Great Tit nestlings alive in the nest at 11 days of age. This absence of a year effect may have been due to less variation across the years in numbers of young indicating consistent, and relatively poor, performance irrespective of better conditions in some springs. Although characteristics of the tree canopy were dominant in most models for both species, other variables were also significant and especially so for Blue Tits. Availability of margin/patch habitat was the only habitat variable included in the models for the number of young fledged and overall success for Blue Tits. These were also two of the relatively few relationships for which the best models were derived from a sample area around the nest box with a radius of 50 m rather 100 m, suggesting that proximity/extent of the ES habitat was advantageous for Blue Tits. In woodland, average foraging distances for tits feeding young are typically about 25-50 m with longer distances as foraging habitat quality declines (Naef-Daenzer 2000, Tremblay *et al.* 2005). The selection of 100 m rather 50 m in most of the models was consistent with the lower quality of farmland as foraging habitat, again suggesting that the proximity of ES habitat was favourable. Foraging distances along the linear habitat provided by hedges were likely to exceed 100 m; anecdotal evidence being provided by the remains of peanuts in some nests at least 500 m from the nearest garden.

In contrast to this effect of ES habitat for Blue Tits, the margin/patch proximity index was negative for total live biomass of young at 11 days of age for Great Tits (Table 9.16), whereas it was positive for clutch size. This might suggest that foraging in such habitat did not provide Great Tits with an effective return for the time and effort expended when feeding young, but did provide laying females with a useful source of food. Such relationships may be influenced by travel costs, i.e. whilst egg laying, the female is largely free to move between foraging sites and not constrained to return food to the nest (e.g. Hinsley 2000), but in large part this is only speculation. The only other negative variable for Great Tits, hedge volume, also occurred in the model for clutch size and could have indicated that hedges with fewer trees were less favourable for this species. For Blue Tits,

the only negative variable, tree canopy volume, occurred in the model for mean nestling body mass and was hard to interpret given that the only other, and positive, habitat variable in the model was tree canopy proximity index. It is possible that this was a statistical artefact due to the relationship between these two variables.

The selection of non-tree variables in two of the Blue Tits models, versus their appearance in all of them for Great Tits, suggested that the former species might be more flexible in its use of alternative habitat types. Similarly, habitat variables influenced timing of breeding in Great Tits, but not in Blue Tits. Previous work has also shown that Blue Tits appear to be more resilient to changes in woodland habitat and variation in habitat structure (Hinsley *et al.* 1999, Hinsley *et al.* 2002, Mackenzie unpubl. data). The fact that both the negative variables in the Great Tit models involved non-tree habitats whereas the one negative variable for Blue Tits related to trees was also consistent with this observation. This raises the possibility that Great Tit breeding success might provide a measure of tree-related habitat features on farmland whereas Blue Tit performance might provide information about alternative habitats, perhaps especially in landscapes largely lacking mature hedgerow trees. The increase in the proportion of boxes occupied by Great Tits after 2007, and the concomitant decrease in Blue Tits, is not an effect unique to farmland. In the first year that nest boxes (accessible to both species) are provided for tits in any habitat, more or at least as many are occupied by Blue Tits and then the proportion used by Great Tits increases in the next year and sometimes also subsequent years. This is a consequence of the social behaviour of the two species and the ability of the larger Great Tit to usually out-compete the smaller species for nest sites which are accessible to both.

9.6 **Conclusions**

● There were large inter-species differences in the responses of birds to habitat provision under ES. Such effects will be driven by species individual ecologies coupled with differences in the primary demographic factors responsible for each species population dynamics (Siriwardena *et al.* 2000, Butler *et al.* 2010, Perkins *et al.* 2011). There is also evidence from other studies that the degree of response to ES provision is influenced by larger-scale effects resulting in greater benefits in more diverse landscapes supporting greater biodiversity (Whittingham 2011). This is not surprising (recovery from rock bottom is always likely to be

the toughest prospect) but makes the point that ES provision may need to be tailored to locations as well as species.

● There was evidence that the scale of winter habitat use differed between bird species, probably in relation to the availability of their preferred resources in the landscape. For example, the apparent movement of Yellowhammers out of the study area in winter, and the sporadic occurrence of Reed Buntings.

● Providing winter seed benefited the birds but effects may have been species-specific depending on the type, especially size and availability, of seed provided. In terms of effects on winter bird abundance at a whole-farm scale, the Hillesden patches were of most benefit to Linnets and Chaffinches. The provision of more, larger-seeded plant species, e.g. cereals, may be required to benefit Yellowhammers, Reed Buntings and perhaps Greenfinches. Chicory (present in the patches in 2006/07 as a contaminant) was of benefit to Goldfinches in the first half of the winter and they also used teasels in the tussocky margins.

● The benefits of winter bird food patches were not confined to the provision of seed; depending on the species sown, patches also provided cover, a modified microclimate, e.g. access to damp ground, and invertebrate food supplies. These attributes were thought to be of particular benefit to Dunnocks and Song Thrushes.

● To maximise cost-effectiveness, patches need to be managed to be as productive as possible. Increasing seed availability will increase bird numbers in winter (at least in terms of the range of numbers present at Hillesden); thus increases in both patch quality and patch size should be beneficial.

● Bigger, more productive patches may help meet the “hungry gap”, i.e. supplies will last longer and may also help conserve “natural” food supplies for longer into the winter.

● Supplemental feeding may also address the “hungry gap”, but this was not tested directly at Hillesden, and to cater for more than just the large-billed buntings and finches, would require a range of seed sizes to be supplied.

● There was little evidence of bird use of non-patch ES habitat in late winter – but such habitat could be used at other times (e.g. positive relationships between numbers of territories and the availability of margin and patch habitat for Blackbird, Song Thrush and Whitethroat in 2009), and could have its winter quality increased by increasing the incidence of seed bearing species.

● There was evidence that increases in bird numbers in the winter could translate into more breeding territories. At Hillesden this was most evident in the long-term for Linnets (and for several other species in the short-term).

● Increases in the breeding population will be dependent on the availability of suitable nesting habitat in addition to foraging opportunity. For most lowland arable farmland passerines (Skylark being the notable exception), nesting habitat will usually be provided by some form of hedgerow and its associated bottom/edge vegetation, the particular characteristics of the hedge differing between species.

● The increase in Linnets in both seasons may have been due to a combination of suitable seeds supplied by the sown species in the patches, the birds' ability to also utilise the weed species present in the patches and the availability of good quality hedges for breeding.

● There was evidence of time lags in the responses of the birds to different Treatments in both seasons, but most clearly during the breeding season. Such responses showed a trend towards greater numbers (of territories in the breeding season and birds in the winter) in ELS-X and ELS.

● Winter weather was a big modifier of bird responses at both a Treatment scale and a whole farm scale.

● The lack of evidence of a Treatment effect on breeding productivity might have indicated a lack of sensitivity in the method or that such effects need to be examined at the level of individual territories and/or to consider likely foraging distances. Brood size and quality of young may also need to be considered.

● Given the differences in scope and methodology it is difficult to make precise comparisons between the changes in bird numbers at Hillesden and national population trends over the same time period, but overall, changes at Hillesden were broadly similar to, or more favourable, than national trends.

● Tit breeding success was poor on farmland compared to that achieved in woodland, probably as a consequence of reduced food supplies and longer foraging distances.

● Exposure to poor weather during the breeding season was an important additional factor in reduced success.

● Compared to Great Tits, Blue Tits made more use of habitats other than the tree canopy when rearing young.

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11 References

Aebischer, N.J. (1991) Twenty years of monitoring invertebrates and weeds in cereal fields in Sussex. In: Firbank, L.G., Carter, N., Darbyshire, J.F. & Potts, G.R. (eds) The Ecology of Temperate Cereal Fields: 305-331. Blackwell Scientific Publications, Oxford.

Alexander L.V. & Jones P.D. (2001). Updated Precipitation Series for the U.K. and Discussion of Recent Extremes. Atmospheric Science Letters, 1, 142-150.

Altieri M.A., Letourneau D.K. & Risch S.J. (1984). Vegetation diversity and insect pest outbreaks. Critical Reviews in Plant Sciences, 2, 131-169.

Andow D. (1983). The extent of monoculture and its effects on insect pest populations with particular reference to wheat and cotton. Agriculture, ecosystems & environment, 9, 25-35.

Askew, N.P., Searle, J.B., and Moore, N.P. 2007. Agri-environment schemes and foraging of barn owls Tyto alba. Agriculture, Ecosystems & Environment, 118:109-114.

Baker R.R. & Sadovy Y. (1978). The distance and nature of the light-trap response of moths.

Baker R.R. (1985). Moths: Population estimates, light-traps and migration. Case studies in population biology. Manchester Univ. Press, Manchester, 188-211.

Benton T.G., Vickery J.A. & Wilson J.D. (2003). Farmland biodiversity: is habitat heterogeneity the key? Trends in Ecology & Evolution, 18, 182-188.

Bibby, C.J., Hill, D.A., Burgess, N.D. & Mustoe, S. 2005. Bird census techniques. Academic Press, London.

Biesmeijer, J.C., Roberts, S.P.M., Reemer, M., Ohlemuller, R., Edwards, M., Peeters, T., Schaffers, A.P., Potts, S.G., Kleukers, R., Thomas, C.D., Settele, J., & Kunin, W.E. (2006) Parallel declines in pollinators and insect-pollinated plants in Britain and the Netherlands. Science, 313, 351-354.

Birkinshaw N. & Thomas C.D. (1999). Torch-light transect surveys for moths. Journal of Insect Conservation, 3, 15-24.

Blumenshine S.C., Lodge D.M. & Hodgson J.R. (2000). Gradient of fish predation alters body size distributions of lake benthos. Ecology, 81, 374-386.

Boatman, N.D., Stoate, C. & Watts, P.N. 2000. Practical management solutions for birds on lowland arable farmland. In Aebischer, N.J., Evans, A.D., Grice, P.V. & Vickery, J.A. (eds) Ecology and Conservation of Lowland Farmland Birds: 105-114. British Ornithologists' Union, Tring.

Boatman, N.D. & Stoate, C. 2002. Growing crops to provide food for seed-eating farmland birds in winter. Aspects of Appled Biology 67: 229-236.

Boatman, N.D., Brickle, N.W., Hart, J.D., Holland, J.M., Milsom, T.P., Morris, A.J.,Murray, A.W.A., Murray, K.A. & Robertson, P.A. (2004) Evidence for the indirect effects of pesticides on farmland birds. Ibis 146 (supplement 2), 131-143.

Boatman, N., Ramwell, C., Parry, H., Jones, N., Bishop, J., Gaskell, P., Short, C., Mills, J., and Dwyer, J. 2008. A review of environmental benefits supplied by agri-environment schemes. Land Use Policy Group report FST20/79/041, York.

Bommarco, R., Biesmeijer, J.C., Meyer, B., Potts, S.G., Pöyry, J., Roberts, S.P.M., Steffan-Dewenter, I., & Öckinger, E. (2010) Dispersal capacity and diet breadth modify the response of wild bees to habitat loss. Proceedings of The Royal Society of London, Series B., 277: 2075-2082.

Botham M.S., Brereton T., Middlebrook I., Cruickshanks K.L. & Roy D. (2008). United Kingdom Butterfly Monitoring Scheme Annual Report 2007. In. Centre for Ecology and Hydrology Wallingford.

Botham M.S., Brereton T., Middlebrook I., Cruickshanks K.L., Harrower C., Beckmann B. & Roy D. (2009). United Kingdom Butterfly Monitoring Scheme Annual Report 2008. In. Centre for Ecology and Hydrology Wallingford.

Botham M.S., Brereton T.M., Middlebrook I., Cruickshanks K.L., Zannese A. & Roy D.B. (2010). United Kingdom Butterfly Monitoring Scheme report for 2009. . In. Centre for Ecology and Hydrology Wallingford.

Bradley J.D., Tremewan W.G., Smith A. & Hargreaves B. (1979). British tortricoid moths. Ray Society.

Braganca M.A.L., Zanuncio J.C., Picanco M. & Laranjeiro A.J. (1998). Effects of environmental heterogeneity on Lepidoptera and Hymenoptera populations in Eucalyptus plantations in Brazil. Forest Ecology and Management, 103, 287-292.

Branquart E. & Hemptinne J.L. (2000). Selectivity in the exploitation of floral resources by hoverflies (Diptera : Syrphinae). Ecography, 23, 732-742.

Brickle, N.W., Harper, D.G.C., Aebischer, N.J., Cockayne, S.J., (2000). Effects of agriculturalintensification on the breeding success of corn buntings Miliariacalandra. Journal of Applied Ecology 37, 742–756.

Browne, S.J., and Aebischer, N.J. (2003). Arable Stewardship: Impact of the Pilot Scheme on the Brown Hare and Grey Partridge after Five Years. Report to Defra (Contract ref. RMP1870vs3).

Buckingham, D.L. & Peach, W.J. (2006). Leaving final-cut grass sileage in situ overwinter as a seed resource for declining farmland birds. Biodiversity and Conservation 15: 3827-3845.

Butler S.J., Vickery J.A., Norris K. (2007) Farmland Biodiversity and the Footprint of Agriculture. Science 315(5810), 381-384. (doi:10.1126/science.1136607).

Butler, S.J., Mattison, E.H.A., Glithero, N.J., Robinson, L.J., Atkinson, P.W., Gillings, S., Vickery, J.A. & Norris, K. (2010). Resource availability and the persistence of seed-eating bird populations in agricultural landscapes: a mechanistic modelling approach. Journal of Applied Ecology 47: 67-75.

Carvell, C., D. B. Roy, S. M. Smart, R. F. Pywell, C. D. Preston, and D. Goulson. (2006). Declines in forage availability for bumblebees at a national scale. Biological Conservation 132:481-489.

Carvell C., Meek W.R., Pywell R.F., Goulson D. & Nowakowski M. (2007). Comparing the efficacy of agri-environment schemes to enhance bumble bee abundance and diversity on arable field margins. J. appl. Ecol., 44, 29-40.

Chamberlain D.E., Fuller R.J., Bunce R.G.H., Duckworth J.C. & Shrubb M. (2000). Changes in the abundance of farmland birds in relation to the timing of agricultural intensification in England and Wales. Journal of Applied Ecology, 37, 771-788.

Chapman J.W., Reynolds D.R., Smith A.D., Riley J.R., Pedgley D.E. & Woiwod I.P. (2002). High-altitude migration of the diamondback moth Plutella xylostella to the UK: a study using radar, aerial netting, and ground trapping. Ecological Entomology, 27, 641-650.

Conrad K.F., Warren M.S., Fox R., Parsons M.S. & Woiwod I.P. (2006). Rapid declines of common, widespread British moths provide evidence of an insect biodiversity crisis. Biological Conservation, 132, 279-291.

Corbet, S. A., I. H. Williams, and J. L. Osborne. (1991). Bees and the pollination of crops and wild flowers in the European Community. Bee World 72:47-59.

Cowie, R.J. & Hinsley, S.A. (1987). Breeding success of Blue Tits (Parus caeruleus) and Great Tits (Parus major) in suburban gardens. Ardea 75: 81-90.

Davey C.M., Vickery J.A., Boatman N.D., Chamberlain D.E., Parry H.R., Siriwardena G.M. 2010 Assessing the impact of Entry Level Stewardship on lowland farmland birds in England. Ibis 152(3), 459-474.

Dhondt, A.A., Kempenaers, B. & De Laet, J. (1991). Protected winter roosting sites as a limiting resource for blue tits. Acta XX Congressus Internationalis Ornithologici, pp. 1436-1443.

Diaz, M. 1990. Interspecific patterns of seed selection among granivorous passerines: effects of seed size, seed nutritive value and bird morphology. Ibis 132: 467-476.

Doak P. (2000). Population consequences of restricted dispersal for an insect herbivore in a subdivided habitat. Ecology, 81, 1828-1841.

Douglas, D.J.T., Vickery, J.A. & Benton, T.G. (2009). Improving the value of field margins as foraging habitat for farmland birds. Journal of Applied Ecology 46: 353-362.

Driver, E.A. (1981) Calorific values of pond invertebrates eaten by ducks. Freshwater Biology, 11, 579-581

Dudley R. (2002). The Biomechanics of Insect Flight: Form, Function, Evolution. Princeton University Press, Princeton, New Jersey.

Dudley, R. (2002) The Biomechanics of Insect Flight: Form, Function, Evolution. Princeton University Press, Princeton, New Jersey.

Ekroos, J., Heliola, J., & Kuussaari, M. (2010) Homogenization of lepidopteran communities in intensively cultivated agricultural landscapes. Journal of Applied Ecology, 47: 459-467.

Ewers R.M. & Didham R.K. (2006). Confounding factors in the detection of species responses to habitat fragmentation. Biological Reviews, 81, 117-142.

Field, R.H., Morris, A.J., Grice, P.V. & Cooke, A. 2009. Winter use of seed-bearing crops by birds within the English Environmental Stewardship scheme. Abstract only. http://bouproc.blogspot.com/2010/06/lowland-farmland-birds-3-abstract-field.html

Forup M.L. & Memmott J. (2005). The restoration of plant-pollinator interactions in hay meadows. Restoration Ecology, 13, 265-274.

Fox R., Conrad K.F., Parsons M.S., Warren M.S. & Woiwod I.P. (2006). The state of Britain's larger moths. Butterfly Conservation and Rothampstead Research, Wareham, Dorset, 33pp.

Fox R., Brereton T.M., Asher J., Botham M.S., Middlebrook I., Roy D.B. & Warren M.S. (2011). The State of the UK's Butterflies 2011. In. Butterfly Conservation and the Centre for Ecology & Hydrology, Wareham, Dorset.

Fuentes-Montemayor E., Goulson D. & Park K.J. (2011). The effectiveness of agri-environment schemes for the conservation of farmland moths: assessing the importance of a landscape-scale management approach. Journal of Applied Ecology.

Fuller R.J., Gregory R.D., Gibbons D.W., Marchant J.H., Wilson J.D., Baillie S.R. & Carter N. (1995). Population declines and range contractions among lowland farmland birds in Britain. Conservation Biology, 9, 1425-1441.

Gallai, N., Salles, J., Settele, J., & Vaissière, B.E. (2009) Economic valuation of the vulnerability of world agriculture confronted to pollinator decline. Ecological Economics, 68: 810-821.

Garibaldi, L. A., I. Steffan-Dewenter, C. Kremen, et al. (2011). Stability of pollination services decreases with isolation from natural areas despite honey bee visits Ecology Letters 14:1062–1072.

Gibbons,D.W., Bohan,D.A., Rothery,P, Stuart,R., Haughton,A.J., Scott,R., Wilson,J.D., Perry, J.N., Clark,S.J., Dawson, R.J.G, Firbank, L.G. (2006) Weed seed resources for birds in fields with contrasting conventional and genetically modified herbicide-tolerant crops. Proceedings of the Royal Society B, 273 1921-1928.

Gilroy, J.J., Anderson, G.Q.A., Grice, P.V., Vickery, J.A., Bray, I., Watts, P.N. & Sutherland, W.J. (2008). Could soil degradation contribute to farmland bird declines? Links between soil penetrability and the abundance of Yellow Wagtails Motacilla flava in arable fields. Biological Conservation 141: 3116-3126.

Green R.E., Cornell S.J., Scharlemann J.P.W. & Balmford A. (2005). Farming and the Fate of Wild Nature. Science, 307, 550-555.

Greenleaf S.G., Williams N.M., Winfree R. & Kremen C. (2007). Bee foraging ranges and their relationship to body size. Oecologia, 153, 589-596.

Gregory, R.D., Noble, D.G.& Custance, J. 2004. The state of play of farmland birds: population trends and conservation status of lowland farmland birds in the United Kingdom. Ibis 146 (Supple. 2): 1-13.

Gustafson, E.J. & Parker, G.R. (1992) Relationships between landcover proportions and indices of landscape spatial pattern. Landscape Ecology 7: 101-110.

Hanski I. & Ovaskainen O. (2000). The metapopulation capacity of a fragmented landscape. Nature, 404, 755-758.

Harestad A.S. & Bunnell F.L. (1979). Home range and body weight-A re-evaluation. Ecology 60, 389-482.

Haskell J.P., Ritchie M.E. & Olff H. (2002). Fractal geometry predicts varying body size scaling relationships for mammal and bird home ranges. Nature, 418, 527-530.

Heard M.S. (2003). Non-crop plants in fields with contrasting conventional and GM herbicide-tolerant crops. 1. Main effects of treatments. Philosophical Transactions B.

Heard, M.S., Carvell, C., Carreck, N.L., Rothery, P., Osborne, J.L., & Bourke, A.F.G. (2007) Landscape context not patch size determines bumble-bee density on flower mixtures sown for agri-environment schemes. Biology Letters, 3, 638–641.

Henderson, I.G., Vickery, J.A. & Carter, N. (2004) The use of winter bird crops by farmland birds in lowland England. Biological Conservation 118: 21-32.

Hinsley, S.A., Rothery, P. & Bellamy, P.E. (1999) Influence of woodland area on breeding success in Great Tits (Parus major) and Blue Tits (Parus caeruleus). Journal of Avian Biology 30: 271-281.

Hinsley, S.A. (2000) The costs of multiple patch use by birds. Landscape Ecology 15: 765-775.

Hinsley, S.A., Hill, R.A., Gaveau, D.L.A. & Bellamy, P.E. (2002) Quantifying woodland structure and habitat quality for birds using airborne laser scanning. Functional Ecology 16: 851-857.

Hinsley, S.A., Hill, R.A., Bellamy P.E. & Balzter, H. (2006) The application of LiDAR in woodland bird ecology: Climate, canopy structure and habitat quality. Photogrammetric Engineering & Remote Sensing 72: 1399-1406.

Hinsley, S.A., Hill, R.A., Bellamy, P.E., Harrison, N.M., Speakman, J.R., Wilson, A.K. & Ferns, P.N. (2008) Effects of structural and functional habitat gaps on breeding woodland birds: working harder for less. Landscape Ecology 23: 615-626.

Hinsley, S.A., Hill, R.A., Bellamy, P.E., Broughton, R.K., Harrison, N.M., Mackenzie, J.A., Speakman, J.R. & Ferns, P.N. (2009) Do highly modified landscapes favour generalists at the expense of specialists? An example using woodland birds. Landscape Research 34: 509-526.

Hinsley S.A., Redhead J.W., Bellamy P.E., Broughton R.K., Hill R.A., Heard M.S. & Pywell R.F. (2010a). Testing agri-environment delivery for farmland birds at the farm scale: the Hillesden experiment. Ibis, 152, 500-514.

Hinsley, S.A., Novakowski, M., Heard, M., Bellamy, P.E., Broughton, R.K., Hulmes, S., Hulmes, L., Peyton, J. & Pywell, R. (2010b). Performance and effectiveness of winter bird food patches established under Environmental Stewardship: Results from the Hillesden experiment. In: Agri-environment schemes – what have they achieved and where do we go from here? Boatman, N., Green, M., Holland, J., Marshall, J., Renwick, A., Siriwardena, G., Smith, B. & de Snoo, G. (eds). Association of Applied Biologists Annual Conference, University of Leicester, pp. 151-158, April 2010.

Hof A.R. & Bright P.W. (2010). The impact of grassy field margins on macro-invertebrate abundance in adjacent arable fields. Agriculture, ecosystems & environment, 139, 280-283.

Hof, A.R., and Bright, P.W. (2010). The value of agri-environment schemes for macro-invertebrate feeders: hedgehogs on arable farms in Britain. Animal Conservation, 13:467-473.

Hole, D.G., Whittingham, M.J., Bradbury, R.B., Anderson, G.Q.A., Lee, P.L.M., Wilson, J.D. & Krebs, J.R. (2002) Widespread local House Sparrow extinctions – Agricultural intensification is blamed for the plummeting populations of these birds. Nature, Lond. 418: 931-932.

Holland, J.M., Hutchison, M.A.S., Smith, B. & Aebischer, N.J. (2006). A review of invertebrates and seed-bearing plants as food for farmland birds in Europe. Annals of Applied Biology 148: 49–71.

Holling C.S. (1992). Cross-scale morphology, geometry and dynamics of ecosystems. Ecological Monographs, 62, 447-502.

Holyoak M., Jarosik V. & Novak I. (1997). Weather-induced changes in moth activity bias measurement of long-term population dynamics from light trap samples. Entomologia Experimentalis Et Applicata, 83, 329-335.

Holzschuh A., Steffan-Dewenter I., Kleijn D. & Tscharntke T. (2007). Diversity of flower-visiting bees in cereal fields: effects of farming system, landscape composition and regional context. J. appl. Ecol., 44, 41-49.

Humbert J.-Y., Ghazoul J., Sauter G.J. & Walter T. (2010). Impact of different meadow mowing techniques on field invertebrates. Journal of Applied Entomology, 134, 592-599.

Isaacs R., Tuell J., Fiedler A., Gardiner M. & Landis D. (2009). Maximizing arthropod-mediated ecosystem services in agricultural landscapes: the role of native plants. Frontiers in Ecology and the Environment, 7, 196-203.

Jauker F., Diekotter T., Schwarzbach F. & Wolters V. (2009). Pollinator dispersal in an agricultural matrix: opposing responses of wild bees and hoverflies to landscape structure and distance from main habitat. Landsc. Ecol., 24, 547-555.

Keitt T.H. (2009). Habitat conversion, extinction thresholds, and pollination services in agroecosystems. Ecol. Applic., 19, 1561-1573.

Kendleigh, S.C. Dol'nik, V.R., Gavrilov, V.M. (1977) Avian energetics. In: granivorous birds in ecosystems. J.Pinkowski &S.C. Kendleigh (eds) pp127-204. International biological programme 12, cambridge University press, Cambridge.

Kleijn D. & Sutherland W.J. (2003). How effective are European agri-environment schemes in conserving and promoting biodiversity? Journal of Applied Ecology, 40, 947-969.

Kleijn D., Baquero R.A., Clough Y., Diaz M., De Esteban J., Fernandez F., Gabriel D., Herzog F., Holzschuh A., Johl R., Knop E., Kruess A., Marshall E.J.P., Steffan-Dewenter I., Tscharntke T., Verhulst J., West T.M. & Yela J.L. (2006). Mixed biodiversity benefits of agri-environment schemes in five European countries. Ecology Letters, 9, 243-254.

Klein, A.-M., Vaissiére, B.E., Cane, J.H., Steffan-Dewenter, I., Cunningham, S.A., Kremen, C., & Tscharntke, T. (2007) Importance of pollinators in changing landscapes for world crops. Proceedings of the Royal Society of London B 274: 303-313.

Kleijn, D., and I. Raemakers. (2008) A retrospective analysis of pollen host plant use by stable and declining bumble bee species. Ecology 89:1811-1823.

Knight M.E., Martin A.P., Bishop S., Osborne J.L., Hale R.J., Sanderson A. & Goulson D. (2005). An interspecific comparison of foraging range and nest density of four bumblebee (Bombus) species. Molecular Ecology, 14, 1811-1820.

Knop E. & Herzog F. (2007). Biodiversity benefits from connectedness of restoration meadows. Agrarforschung, 14.

Knop E., Herzog F. & Schmid B. (2007). Effect of connectivity between restoration meadows on invertebrates with contrasting dispersal abilities. Restoration Ecology.

Kremen C., Williams N.M., Aizen M.A., Gemmill-Herren B., LeBuhn G., Minckley R., Packer L., Potts S.G., Roulston T., Steffan-Dewenter I., Vázquez D.P., Winfree R., Adams L., Crone E.E., Greenleaf S.S., Keitt T.H., Klein A.-M., Regetz J. & Ricketts T.H. (2007). Pollination and other ecosystem services produced by mobile organisms: a conceptual framework for the effects of land-use change. Ecology Letters, 10, 299-314.

Kuussaari M., Heliola J., Luoto M. & Poyry J. (2007a). Determinants of local species richness of diurnal Lepidoptera in boreal agricultural landscapes. Agriculture Ecosystems & Environment, 122, 366-376.

Kuussaari M., Heliola J., Poyry J. & Saarinen K. (2007b). Contrasting trends of butterfly species preferring semi-natural grasslands, field margins and forest edges in northern Europe. Journal of Insect Conservation, 11, 351-366.

Lobley, M., Saratsi, E. & Winter, M. (2009) Training and advice for agri-environmental management. http://www.bou.org.uk/bouproc-net/lfb3/lobley-etal.pdf

Lonsdorf, E., Kremen, C., Ricketts, T., Winfree, R., Williams, N. & Greenleaf, S. (2009) Modelling pollination services across agricultural landscapes. Annals of Botany, 103, 1589-1600.

Macdonald, D.W., Tew, T.E., Todd, I.A., Garner, J.P., and Johnson, P.J. (2000) Arable habitat use by wood mice (Apodemus sylvaticus). A farm-scale experiment on the effects of crop rotation. Journal of Zoology, 250:313-320.

Macdonald, D.W., Tattersall, F.H., Service, K.M., Firbank, L.G., and Feber, R.E. (2007) Mammals, agri-environment schemes and set-aside – what are the putative benefits? Mammal Review, 37:259-277.

Marshall E.J.P., West T.M. & Kleijn D. (2006). Impacts of an agri-environment field margin prescription on the flora and fauna of arable farmland in different landscapes. Agriculture, ecosystems & environment, 113, 36-44.

Maudsley, M.J., West, T.M., Rowcliffe, H.R., Marshall, E.J.P., (2000) The impacts of hedge management on wildlife: preliminary results for plants and insects. Aspects of Applied Biology 58, 389–396.

McCleery, R.H. & Perrins, C.M. (1998) Scientific Correspondence, ……..temperature and egg-laying trends. Nature 391: 30-31.

McGeachie W.J. (1988). A remote sensing method for the estimation of light-trap efficiency. Bulletin of Entomological Research, 78, 379-385.

McGeachie W.J. (1989). The effects of moonlight illuminance, temperature and wind speed on light-trap catches of moths. Bulletin of Entomological Research, 79, 185-192.

Merckx T., Feber R.E., Dulieu R.L., Townsend M.C., Parsons M.S., Bourn N.A.D., Riordan P. & MacDonald D.W. (2009a). Effect of field margins on moths depends on species mobility: Field-based evidence for landscape-scale conservation. Agriculture Ecosystems & Environment, 129, 302-309.

Merckx T., Feber R.E., Riordan P., Townsend M.C., Bourn N.A.D., Parsons M.S. & Macdonald D.W. (2009b). Optimizing the biodiversity gain from agri-environment schemes. Agriculture, ecosystems & environment, 130, 177-182.

Merckx T., Feber R.E., McLaughlan C., Bourn N.A.D., Parsons M.S., Townsend M.C., Riordan P. & Macdonald D.W. (2010a). Shelter benefits less mobile moth species: the field-scale effect of hedgerow trees. Agriculture, ecosystems & environment, 138, 147-151.

Merckx T., Feber R.E., Parsons M.S., Bourn N.A.D., Townsend M.C., Riordan P. & Macdonald D.W. (2010b). Habitat preference and mobility of Polia bombycina: are non-tailored agri-environment schemes any good for a rare and localised species? Journal of Insect Conservation, 14, 499-510.

Merckx, T., Berwaerts, K., (2010) What type of hedgerows do Brown hairstreak (Thecla betulae L.) butterflies prefer? Implications for European agricultural landscape conservation. Insect Conservation and Diversity 3, 194–204.

Moorcroft, D., Whittingham, M.J., Bradbury, R.B. & Wilson, J.D. (2002) The selection of stubble fields by wintering granivorous birds reflects vegetation cover and food abundance. Journal of Applied Ecology. 39: 535–47.

Morris, A.J., Wilson, J.D., Whittingham, M.J., Bradbury, R.B., (2005) Indirect effects of pesticides on breeding yellowhammer (Emberiza citrinella). Agriculture, Ecosystems& Environment 106, 1–16.

Naef-Daenzer, B. (2000) Patch time allocation and patch sampling by foraging great and blue tits. Animal Behaviour 59: 989-999.

Nagy, N.E. (1987) Field Metabolic Rate and Food Requirement Scaling in Mammals and Birds.Ecological Monographs 57(2), 111-128.

Natural England (2010). Entry Level Stewardship: Environmental Stewardship Handbook. Natural England, Peterborough.

New T.R. (2004). Moths (Insecta: Lepidoptera) and conservation: background and perspective. Journal of Insect Conservation, 8, 79-94.
Newton I. (2004). The recent declines of farmland bird populations in Britain: an appraisal of causal factors and conservation actions. Ibis, 146, 579-600.
Nieminen M., Rita H. & Uuvana P. (1999). Body size and migration rate in moths. Ecography, 697-707.
Öckinger E. & Smith H.G. (2007). Semi-natural grasslands as population sources for pollinating insects in agricultural landscapes J. Appl. Ecol., 44, 50-59.
Öckinger E., Schweiger O., Crist T.O., Debinski D.M., Krauss J., Kuussaari M., Petersen J.D., Pöyry J., Settele J., Summerville K.S. & Bommarco R. (2010). Life-history traits predict species responses to habitat area and isolation: a cross-continental synthesis. Ecology Letters, 13, 969-979.
Oliver T., Roy D.B., Hill J.K., Brereton T. & Thomas C.D. (2010). Heterogeneous landscapes promote population stability. Ecology Letters, 13, 473-484.
Ollerton, J., Winfree, R., & Tarrant, S. (2011) How many flowering plants are pollinated by animals? Oikos, 120, 321-326.
Osborne J.L., Williams I.H. & Corbet S.A. (1991). Bees, pollination and habitat change in the European Community. Bee World, 72, 99-116.
Osborne, J.L., Martin, A.P., Carreck, N.L., Swain, J.L., Knight, M.E., Goulson, D., Hale, R.J., & Sanderson, R.A. (2008) Bumblebee flight distances in relation to the forage landscape. Journal of Animal Ecology, 77: 406-415.
Parker D.E., Legg T.P. & Folland C.K. (1992). A new daily Central England Temperature Series, 1772-1991. Int. J. Clim., 12, 317-342.
Peach, W.J., Denny, M., Cotton, P.A., Hill, I.F., Gruar, D., Barritt, D., Impey, A. & Mallord, J. (2004) Habitat selection by Song Thrushes in stable and declining farmland populations. Journal of Applied Ecology 41: 275-293.
Perkins, A.J., Anderson, G. & Wilson, J.D. (2007) Seed food preferences of granivorous farmland passerines. Bird Study 54: 46-53.
Perkins, A.J., Maggs, H.E., Watson, A. & Wilson, J.D. 2011. Adaptive management and targeting of agri-environment schemes does benefit biodiversity: a case study of the corn bunting Emberiza calandra. Journal of Applied Ecology 48: 514-522.
Perrins, C.M. (1970). The timing of birds' breeding season. Ibis 112: 242-255.
Potts, S.G., Biesmeijer, J.C., Kremen, C., Neumann, P., Schweiger, O., & Kunin, W.E. (2010) Global pollinator declines: trends, impacts and drivers. Trends in Ecology & Evolution, 25, 345-353.
Preston C.D., Pearman D.A. & Dines T.D. (eds.) (2002). New atlas of the British and Irish Flora. Oxford University Press, Oxford.
Prŷs-Jones, R. (2002). Reed Bunting, Emberiza schoeniclus. In Wernham, C.V., Toms, M.P., Marchant, J.H., Clark, J.A., Siriwardena, G.M. & Baille, S.R. (eds) The Migration Atlas: movements of the birds of Britain and Ireland: 675-677. T. & A.D. Poyser, London.
Pywell R.F., Warman E.A., Hulmes L., Hulmes S., Nuttall P., Sparks T.H., Critchley C.N.R. & Sherwood A. (2006). Effectiveness of new agri-environment schemes in providing foraging resources for bumblebees in intensively farmed landscapes. Biol. Conserv., 129, 192-206.
R Development Core Team. (2009) R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, URL http://www.R-project.org.

Reid, N., McDonald, R.A., and Montgomery, W.I. (2007) Mammals and agri-environment schemes: hare haven or pest paradise? Journal of Applied Ecology, 44:1200-1208.

Ricketts T.H., Daily G.C., Ehrlich P.R. & Fay J.P. (2001). Countryside biogeography of moths in a fragmented landscape: biodiversity in native and agricultural habitats. Conservation Biology, 15, 378-388.

Ricketts T.H., James R., Ingolf S.-D., Saul A.C., Claire K., Anne B., Barbara G.-H., Sarah S.G., Alexandra M.K., Margaret M.M., Lora A.M., Alfred O. & Blande F.V. (2008). Landscape effects on crop pollination services: are there general patterns? Ecology Letters, 11, 499-515.

Riddington, R. & Gosler, A.G. (1995) Differences in reproductive success and parental qualities between habitats in the Great Tit Parus major. Ibis 137: 371-378.

Risely, K., Noble, D.G. & Baillie, S.R. (2008) The Breeding Bird Survey 2007. BTO Research Report 508. British Trust for Ornithology, Thetford.

Risely, K., Noble, D.G. & Baillie, S.R. (2009) The Breeding Bird Survey 2008. BTO Research Report 537. British Trust for Ornithology, Thetford.

Risely, K., Baillie, S.R., Eaton, M.A., Joys, A.C., Musgrove, A.J. Noble, D.G., Renwick, A.R. & Wright, L.J. (2010). The Breeding Bird Survey 2009. BTO Research Report 559. British Trust for Ornithology, Thetford.

Risely, K., Renwick, A.R., Dadam, D., Eaton, M.A., Johnston, A., Baille, S.R., Musgrove, A.J., & Noble, D.G. (2011) The Breeding Bird Survey 2010. BTO Research Report 597. British Trust for Ornithology, Thetford.

Robinson H.S. & Robinson P.J.M. (1950). Some notes on the observed behaviour of Lepidoptera in flight in the vicinity of light- sources together with a description of a light-trap designed to take entomological samples. Entomologist's Gazette 1, 3-20.

Robinson R.A. & Sutherland W.J. (2002). Post-war changes in arable farming and biodiversity in Great Britain. Journal of Applied Ecology, 39, 157-176.

Robinson, R.A., Hart, J.D., Holland, J.M. & Parrott, D. (2004) Habitat use by seed-eating birds: a scale-dependent approach. Ibis 146 (Suppl. 2): 87-98.

Rogers L.E., Hinds W.T. & Buschbom R.L. (1976). A general weight vs. length relationship for insects. Ann. Entomol. Soc. Am., 69, 387-389.

Rundlöf M., Nilsson H. & Smith H.G. (2008). Interacting effects of farming practice and landscape context on bumble bees. Biological Conservation, 141, 417-426.

Schabenberger O. & Pierce F.J. (2002). Contemporary Statistical Models for the Plant and Soil Sciences. CRC Press, London.

Shore, R.F., Meek, W.R., Sparks, T.H., Pywell, R.F., and Nowakowski, M. (2005) Will Environmental Stewardship enhance small mammals abundance on intensively managed farmland? Mammal Review, 35:277-284.

Siriwardena G.M., Baillie S.R., Buckland S.T., Fewster R.M., Marchant J.H. & Wilson J.D. (1998). Trends in abundance of farmland birds: a quantitative comparison of smoothed Common Birds Census indices. Journal of Applied Ecology, 35, 24-43.

Siriwardena, G.M., Baillie, S.R., Crick, H.Q.P., Wilson, J.D. & Gates, S. (2000) The demography of lowland farmland birds. In Aebischer, N.J., Evans, A.D., Grice, P.V. & Vickery, J.A. (eds) Ecology and Conservation of Lowland Farmland Birds: 117-133. Tring: British Ornithologists' Union.

Siriwardena, G.M. & Stevens, D.K. (2004) Effects of habitat on the use of supplementary food by farmland birds in winter. Ibis 146 (Suppl. 2): 144-154.

Siriwardena, G.M., Calbrade, N.A., Vickery, J.A. & Sutherland, W.J. (2006) The effect of the spatial distribution of winter seed food resources on their use by farmland birds. Journal of Applied Ecology 43: 628-639.

Siriwardena, G.M. & Anderson, G.Q.A. (2007) How can agri-environment measures providing winter food for birds best deliver population increases? Aspects of Applied Biology 81: 117-126.

Siriwardena, G.M., Calbrade, N.A. & Vickery, J.A. (2008) Farmland birds and late winter food: does seed supply fail to meet demand? Ibis 150: 585-595.

Siriwardena, G.M. (2010) The importance of spatial and temporal scale for agri-environment scheme delivery. Ibis: 152: 515-529.

Snow, D., Snow, B.K., (1988) Birds and Berries: A Study of an Ecological Interaction. T. & A.D. Poyser, Carton.

Southwood, T.R.E. & Cross, D.J. (2002) Food requirements of grey partridge Perdix perdix chicks. - Wildlife Biology,. 8: 175-183

Spalding A. (1997). The use of the butterfly transect method for the study of the nocturnal moth Luperina nickerlii leechi Goater (Lepidoptera: Noctuidae) and its possible application to other species. Biological Conservation, 80, 147-152.

Staley J.T., Sparks T.H., Croxton P.J., Baldock K.C.R., Heard M.S., Hulmes S., Hulmes L., Peyton J., Amy S.R., Pywell R.F. (2012) Long-term effects of hedgerow management policies on resource provision for wildlife. Biological Conservation 145(1), 24-29.

Steffan-Dewenter I. & Tscharntke T. (2000). Butterfly community structure in fragmented habitats. Ecology Letters, 3, 449-456.

Steffan-Dewenter I. & Tscharntke T. (2002). Insect communities and biotic interactions on fragmented calcareous grasslands - a mini review. Biological Conservation, 104, 275-284.

Steffan-Dewenter I., Munzenberg U., Burger C., Thies C. & Tscharntke T. (2002). Scale-dependent effects of landscape context on three pollinator guilds. Ecology, 83, 1421-1432.

Steffan-Dewenter I. & Kuhn A. (2003). Honeybee foraging in differentially structured landscapes. Proc. R. Soc. Lond. B, Biol. Sci., 270, 569-575.

Stevens V.M., Turlure C. & Baguette M. (2010). A meta-analysis of dispersal in butterflies. Biological Reviews, 85, 625-642.

Stoate, C., Henderson, I.G. & Parish, D.M.B. (2004) Development of an agri-environment scheme option: seed bearing crops for farmland birds. Ibis 146 (Supple. 2): 203-209.

Summerville K.S. & Crist T.O. (2004). Contrasting effects of habitat quantity and quality on moth communities in fragmented landscapes. Ecography, 27, 3-12.

Tattersall, F.H., Hart, B.J., Manley, W.J., Macdonald, D.W., and Feber, R.E. (1999a). Small mammals on set-aside blocks and margins. Aspects of Applied Biology, 54:131-138.

Tattersall, F.H., Hart, B.J., Manley, W.J., Macdonald, D.W., and Feber, R.E. (1999b) Does the method of set-aside establishment affect its use by wood mice? Journal of Zoology (London), 249:472-476.

Tew, T.E., Macdonald, D.W., and Rands, M.R.W. (1992) Herbicide application affects microhabitat use by arable wood mice (Apodemus sylvaticus). Journal of Applied Ecology, 29:532-539.

Thomas C.D. (2000). Dispersal and extinction in fragmented landscapes. Proceedings of the Royal Society of London Series B-Biological Sciences, 267, 139-145.

Thomas K., Davis T. & Parsons M. (2006). Update on UK Biodiversity Action Plan moths - a summary for the year 2005. In.

Tilman D., Fargione J., Wolff B., D'Antonio C., Dobson A., Howarth R., Schindler D., Schlesinger W.H., Simberloff D. & Swackhamer D. (2001). Forecasting agriculturally driven global environmental change. Science, N.Y., 292, 281-284.

Tinbergen, J.M. & Boerlijst, M.C. (1990) Nestling weight and survival in individual great tits (Parus major). Journal of Animal Ecology 59: 1113-1127.

Tremblay, I., Thomas, D.W., Blondel, J., Perret, P. & Lambrechts, M.M. (2005) The effect of habitat quality on foraging patterns, provisioning rate and nestling growth in Corsican Blue Tits Parus caeruleus. Ibis 147: 17-24.

Tscharntke T., Klein A.M., Kruess A., Steffan-Dewenter I. & Thies C. (2005). Landscape perspectives on agricultural intensification and biodiversity - ecosystem service management. Ecology Letters, 8, 857-874.

van Langevelde F., Ettema J.A., Donners M., WallisDeVries M.F. & Groenendijk D. (2011). Effect of spectral composition of artificial light on the attraction of moths. Biological Conservation.

Verhulst, S., van Balen, J.H. & Tinbergen, J.M. (1995) Seasonal decline in reproductive success of the Great Tit: variation in time or quality? Ecology 76: 2392-2403.

Vickery, J.A., Evans, A.D., Grice, P.V., Aebischer, N.J. & Brand-Hardy, R. (eds) (2004) Ecology and Conservation of Lowland Farmland Birds II: The Road to Recovery. Ibis 146 (Supple. 2).

Vickery J.A., Chamberlain D., Evans A., Ewing S., Boatman N.D., Pietravalle S., Norris K., Butler S.J. (2008) Predicting the impact of future agricultural change and uptake of Entry Level Stewardship on farmland birds. BTO Research Report No 485 and Appendices.

Waring P. (1980). A comparison of the Heath and Robinson MV moth traps. Entomologistsâ€™ Record and Journal of Variation, 92, 283-289.

Waring P., Townsend M. & Lewington R. (2003). Field guide to the moths of Great Britain and Ireland. British Wildlife Pub.

Weibull A.C., Bengtsson J. & Nohlgren E. (2000). Diversity of butterflies in the agricultural landscape: the role of farming system and landscape heterogeneity. Ecography, 23, 743-750.

Westphal C., Steffan-Dewenter I. & Tscharntke T. (2003). Mass flowering crops enhance pollinator densities at a landscape scale. Ecology Letters, 6, 961-965.

Westphal C., Bommarco R., Carre G., Lamborn E., Morison N., Petanidou T., Potts S.G., Roberts S.P.M., Szentgyorgyi H., Tscheulin T., Vaissiere B.E., Woyciechowski M., Biesmeijer J.C., Kunin W.E., Settele J. & Steffan-Dewenter I. (2008). Measuring bee diversity in different European habitats and biogeographical regions. Ecological Monographs, 78, 653-671.

Westphal, C. et al. (2006) Bumblebees experience landscapes at different spatial scales: possible implications for coexistence. - Oecologia 149: 289-300

Westphal, C., Bommarco, R., Carre, G., Lamborn, E., Morison, N., Petanidou, T., Williams, N.M., Crone, E.E., Roulston, T.H., Minckley, R.L., Packer, L., & Potts, S.G. (2010) Ecological and life-history traits predict bee species responses to environmental disturbances. Biological Conservation, 143: 2280–2291.

Whittingham, M.J. & Evans, K.L. (2004) The effects of habitat structure on predation risk of birds in agricultural landscapes. Ibis 146 (Supple. 2): 210-220.

Whittingham, M.J., Swetnam, R.D., Wilson, J.D., Chamberlain, D.E. & Freckleton, R.P. (2005) Habitat selection by Yellowhammers Emberiza citrinella on lowland farmland at two spatial scales: implications for conservation management. Journal of Applied Ecology 42: 270-280.

Whittingham, M.J. (2011) The future of agri-environment schemes: biodiversity gains and ecosystems service delivery? Journal of Applied Ecology 48: 509-513.

Williams N.M., Crone E.E., Roulston T.H., Minckley R.L., Packer L. & Potts S.G. (2010). Ecological and life-history traits predict bee species responses to environmental disturbances. Biological Conservation, 143, 2280–2291.

Wilson J.D., Morris A.J., Arroyo B.E., Clark S.C. & Bradbury R.B. (1999). A review of the abundance and diversity of invertebrate and plant foods of granivorous birds in northern Europe in relation to agricultural change. Agriculture, ecosystems & environment, 75, 13-30.

Winfree R., Aguilar R., Vázquez D.P., Lebuhn G. & Aizen M.A. (2009). A meta-analysis of bees' responses to anthropogenic disturbance. Ecology, 90, 2068-2076.

Woodcock, B.A., Westbury, D.B., Tscheulin, T., Harrison-Cripps, J., Harris, S.J., Ramsey, A.J., Brown, V.K., Potts, S.G., (2008) Effects of seed mixture and management on beetle assemblages of arable field margins. Agriculture, Ecosystems & Environment 125, 246–254.

Woodcock B.A., Vogiatzakis I.N., Westbury D.B., Lawson C.S., Edwards A.R., Brook A.J., Harris S.J., Lock K.A., Maczey N. & Masters G. (2010). The role of management and landscape context in the restoration of grassland phytophagous beetles. Journal of Applied Ecology, 47, 366-376.

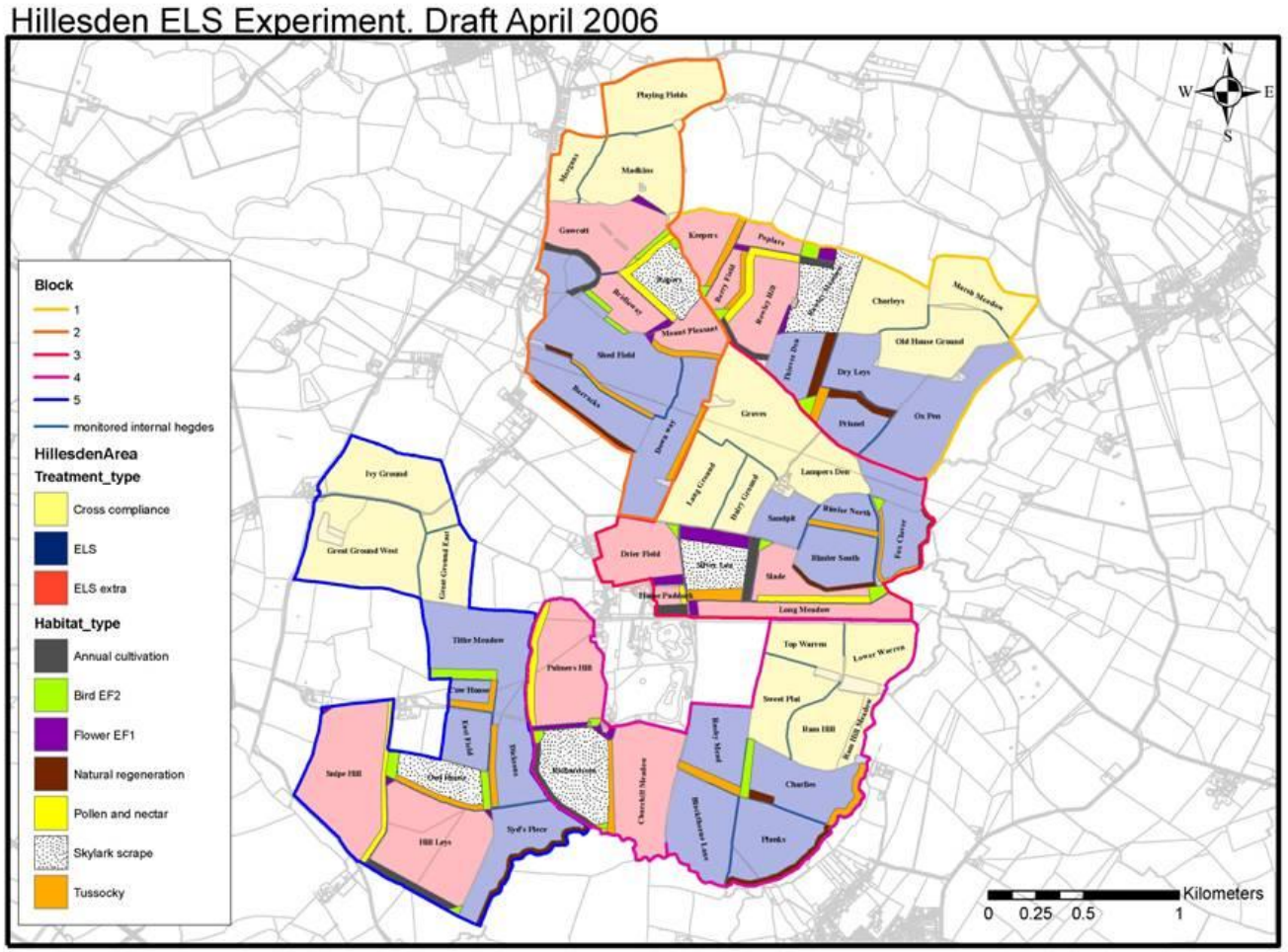
Yela J.L. & Holyoak M. (1997). Effects of moonlight and meteorological factors on light and bait trap catches of noctuid moths (Lepidoptera: Noctuidae). Environmental Entomology, 26, 1283-1290.

Young M. (1997). The natural history of moths. Poyser.

Young M. (2005). Insects in flight. In: Insect Sampling in Forest Ecosystems, pp. 116-145.

12 Appendices

12.1 **Map of Hillesden experimental layout**



12.2

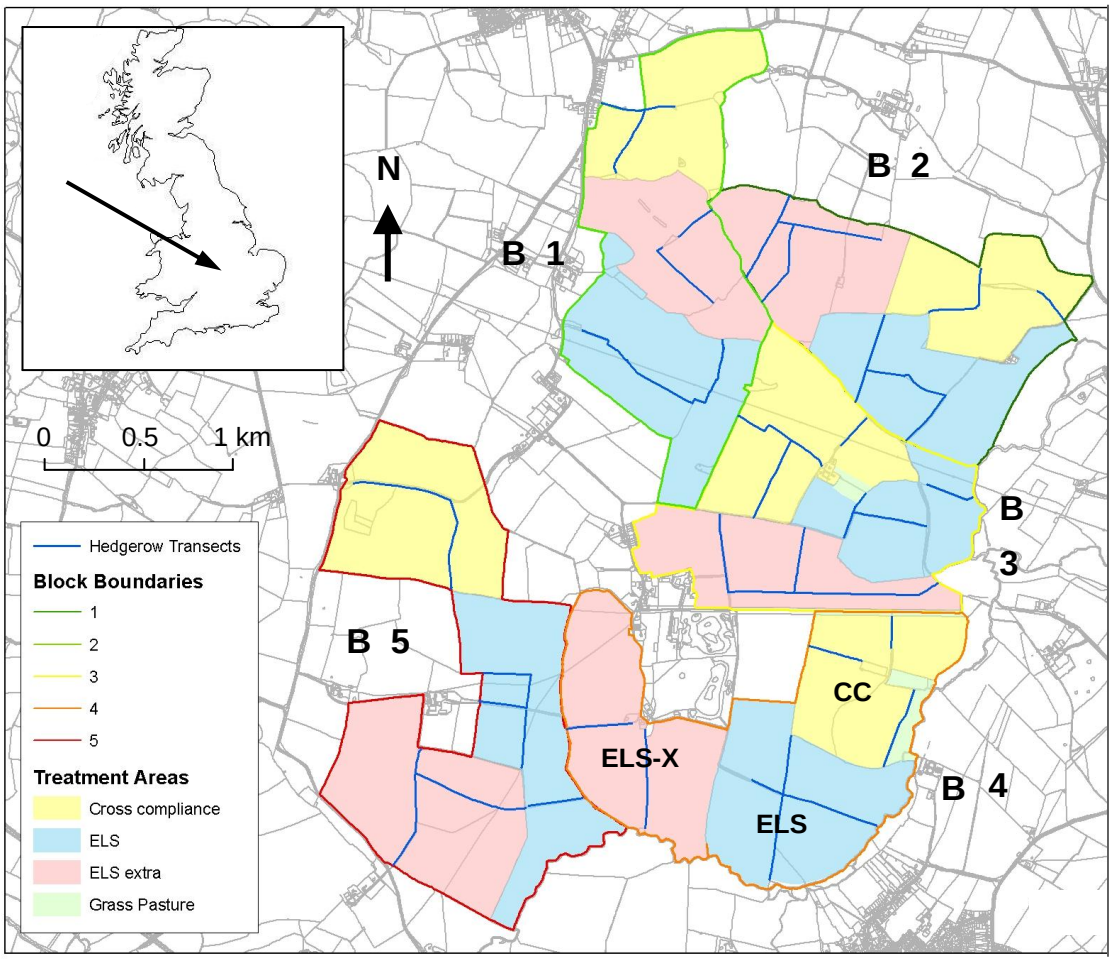
12.3 **Option code and sown plant species list**

Option	Latin name	Common name
EE3	*Dactylis glomerata*	Cock's-foot
EE3	*Festuca arundinacea*	Tall Fescue
EE3	*Festuca pratensis*	Meadow Fescue
EE3	*Festuca rubra*	Red Fescue
EE3+	*Achillea millefolium*	Yarrow
EE3+	*Centaurea nigra*	Common Knapweed
EE3+	*Dactylis glomerata*	Cock's-foot
EE3+	*Daucus carota*	Wild Carrot
EE3+	*Dipsacus fullonum*	Wild Teasel
EE3+	*Festuca arundinacea*	Tall Fescue
EE3+	*Festuca pratensis*	Meadow Fescue
EE3+	*Festuca rubra*	Red Fescue
EE3+	*Lotus corniculatus*	Common Bird's-foot-trefoil
EE3+	*Phleum pratense*	Timothy
EE3+	*Vicia cracca*	Tufted Vetch
EF1	*Achillea millefolium*	Yarrow
EF1	*Centaurea nigra*	Common Knapweed
EF1	*Clinopodium vulgare*	Wild Basil
EF1	*Daucus carota*	Wild Carrot
EF1	*Filipendula ulmaria*	Meadowsweet
EF1	*Galium mollugo*	Hedge Bedstraw
EF1	*Galium verum*	Lady's Bedstraw
EF1	*Knautia arvensis*	Field Scabious
EF1	*Leontodon hispidus*	Rough Hawkbit
EF1	*Leucanthemum vulgare*	Oxeye Daisy
EF1	*Lotus corniculatus*	Common Bird's-foot-trefoil
EF1	*Lychnis flos-cuculi*	Ragged-Robin
EF1	*Malva moschata*	Musk-mallow
EF1	*Plantago media*	Hoary Plantain
EF1	*Primula veris*	Cowslip
EF1	*Prunella vulgaris*	Selfheal
EF1	*Ranunculus acris*	Meadow Buttercup
EF1	*Rumex acetosa*	Common Sorrel
EF1	*Sanguisorba minor*	Salad Burnet
EF1	*Silene dioica*	Red Campion

EF1	*Silene vulgaris*	Bladder Campion
EF1	*Stachys officinalis*	Betony
EF1	*Trifolium pratense*	Red Clover
EF1	*Vicia cracca*	Tufted Vetch
EF1	*Agrostis capillaris*	Common Bent
EF1	*Cynosurus cristatus*	Crested Dog's-tail
EF1	*Festuca rubra*	Red Fescue
EF1	*Festuca rubra*	Slender Creeping Red Fescue
EF2a	*Chenopodium quinoa*	Quinoa
EF2a	*Echinochloa frumentacea*	White Millet
EF2a	*Fagopyrum esculentum*	Buckwheat
EF2a	*Raphanus sativus*	Radish
EF2a	*x Triticosecale*	Triticale
EF2b	*Beta vulgaris*	Beet
EF2b	*Brassica oleracea*	Thousand Head Kale
EF2b	*Chenopodium quinoa*	Quinoa
EF2b	*Cichorium intybus*	Chicory
EF2b	*x Triticosecale*	Triticale
EF2c	*Borago officinalis*	Borage
EF2c	*Chenopodium quinoa*	Quinoa
EF2c	*Echinochloa frumentacea*	White Millet
EF2c	*Helianthus annuus*	Semi-Dwarf Sunflower
EF2c	*Melilotus officinalis*	Sweet Clover
EF2c	*Raphanus sativus*	Radish
EF2c	*x Triticosecale*	Triticale
EF4	*Lotus corniculatus*	Bird's-foot-trefoil
EF4	*Onobrychis viciifolia*	Sainfoin
EF4	*Trifolium hybridum*	Alsike Clover
EF4	*Trifolium pratense*	Red Clover

12.4 **Map of the study area & bird transects**

Map of the study area showing arrangement of the replicate blocks (B1 – B5) and the three treatments (CC, ELS and ELS-X) within each block. Blue lines show the locations of the 15 hedgerow transects. The insert shows the location of the study area in central lowland England.



12.5 Published Paper: "Testing agri-environment delivery for farmland birds at the farm scale: the Hillesden experiment"

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ABSTRACT

The Hillesden experiment, established in 2005/06 to test the delivery of biodiversity benefits under Environmental Stewardship, covers *c.* 1000 ha of arable farmland in central lowland England. It is a randomised block experiment with five replicates of three treatments: i) CC: cross compliance, the control, ii) ELS: 1% of land removed from production for wildlife habitat provision, iii) ELS-X: 5% of land used for wildlife habitat, each treatment being applied to contiguous areas of 70-80 ha. Bird usage of winter food patches, comprising three different seed mixes, was monitored through the winter and was also related to seed yield. Winter and breeding season bird/territory abundance was recorded before and after the provision of the winter food patches. Bird use of the patches differed between seed mixes. There was large variation between individual patches in both seed yield and bird numbers and between individual bird species in their use of different seed mixes, suggesting that the availability of a range of patch types would be beneficial. Use of all patch types declined sharply in late January to February indicating depletion and/or inability of birds to access shed seed. Winter bird abundance at a farm-scale for all species combined, granivorous species and nine individual species increased for all monitored species when seed patches were available. At a treatment level, the increases tended to be greater in ELS-X where most of the patches were located. In the breeding season at a farm-scale, the numbers of territories for all species combined and granivorous species increased significantly when seed patches had been available in the previous winter. There was little evidence of a treatment-scale response. The provision of winter food appeared to increase winter bird abundance and to follow on into an overall increase in the breeding population, but if the latter effect is to be reflected elsewhere, it requires that sufficient breeding habitat is available to accommodate an increase.

INTRODUCTION

Agriculture is the dominant land use in Europe and has a concomitant major impact on biodiversity (Pain & Pienkowski 1997, Schifferli 2000, Robinson & Sutherland 2002,

Donald & Evans 2006, Voříšek *et al.* 2009). Low-intensity agriculture can create fine-grained landscapes with high habitat and structural diversity, but in the last 40-50 years agricultural intensification has been responsible for severe declines in many bird species, especially those most closely associated with cropped land (Fuller *et al.*1995, Chamberlain *et al.* 2000, Fuller 2000, Hole *et al.* 2002, Newton 2004). Agri-environment schemes (AES) comprise a key component of UK Government policy aimed at mitigating habitat loss and degradation in farmland, and specifically to reverse the decline in farmland birds (Swash *et al.* 2000, Bradbury *et al.* 2004, Vickery *et al.* 2004). The Farmland Bird Indicator (FBI; Defra 2009a), one of 18 UK Biodiversity Indicators, averages the population trends of 19 species of farmland birds (both generalists and specialists) as a means of monitoring the effectiveness of measures to improve UK agriculture for biodiversity (Gregory *et al.* 2004). The FBI was adopted in the late 1990s with a Public Service Agreement (PSA) to reverse the declining trend by 2020 (Defra 2008).

Currently, around 63% of agricultural land in England is under some form of agreement at a cost of about £341 m per annum (Defra 2009b). The most recent scheme, Environmental Stewardship (ES), was made available to all farmers in England in 2005, after pilot studies in 2003/04. By the end of 2008, some 5 million hectares were being managed within the Entry Level Scheme and 291 thousand hectares under the Higher Level Scheme (Natural England 2009). Despite these efforts, the overall trend in the Farmland Bird Index is currently downwards. Various reasons related to key ES options (and others, see Discussion) have been suggested for this continued decline, but given the level of uncertainty, there is a clear need to investigate the effectiveness of ES options as they are applied on farmland (Davey *et al.* 2010).

The Hillesden Experiment was designed to evaluate and demonstrate the cost-effectiveness of ES options in conserving and enhancing farmland biodiversity at a farm-scale and to inform future development of ES prescriptions. The work focuses on impacts across a number of taxa including arable weeds, seed/berry production, pollinators (bumble bees, solitary bees and their parasites, Carvell *et al.* 2008), soil micro and macro fauna, butterflies and moths, other invertebrates, small mammals and birds. In this paper, we describe results from the first two years of the five year project detailing bird responses to the provision of winter food patches of several

different seed mixes and the apparent consequences for subsequent numbers of breeding territories. From the results of previous studies (e.g. Boatman *et al.* 2000, Henderson *et al.* 2004, Stoate *et al.* 2004), we hypothesized that food patches would attract greater numbers of birds than cropland, thus increasing bird abundance in winter and providing the potential to enhance the size of the breeding population. We examined responses of both individual species and groups of species and assessed how spatial scale and variation in the data affected our ability to detect differences between treatments compared to changes at a whole-farm scale. We also investigated the effects of patch seed yield on bird usage, with an expectation of a positive relationship, and evaluated the use made by birds of other types of prescribed margin and patch habitat types in late winter.

METHODS

Study site

The experiment is located on about 1000 ha of lowland arable farmland in central England (51°57' N, 1°00' W) near Buckingham (Fig. 1). The site is a typical heavy land farm with a simple rotation of autumn-sown winter wheat, oil-seed rape and field beans. The study was established in 2005/06 and is a randomised block experiment with five replicates of three treatments, each treatment being applied to contiguous areas of 70-80 ha. The three treatments are defined as i) Cross Compliance (CC): annual post-harvest hedge cutting and 6-m wide buffer zones to protect hedges and water courses; this is the control treatment reflecting minimum farming environmental requirements, ii) Entry-Level Scheme (ELS): 1% of land removed from production to create a small number of simple options, namely one winter bird food patch and some grass margins (6 - 8 m), with hedges cut every two years, and iii) Entry-Level Scheme Extra (ELS-X): 5% of land removed from production to create a more diverse range of options including, three sorts of bird food patch and a range of grass and flower margins (6 - 8 m) and patches, plus biennial hedge cutting.

In August 2007, airborne remote sensed data (Light Detection and Ranging, LiDAR, and hyperspectral data) were acquired for an area of *c.* 21 km^2 centred on the study

site. These data have been used to construct a high resolution digital canopy height model (LiDAR) and a land-use map (hyperspectral), both with a pixel size of 0.5 x 0.5 m. Further details of the approach and methodology can be found in Hinsley *et al.* (2002), Hill and Thomson (2005), Hinsley *et al.* (2008) and Hill and Broughton (2009). The canopy height model was used to extract structural data describing hedgerow and tree characteristics (see below) and the land-use map is referred to later in the Discussion in relation to future work.

Winter bird food patches

The winter bird food patches were first sown in the spring of 2006, with one *c.* 0.25 ha patch in each ELS Treatment and three *c.* 0.5 ha patches in each ELS-X Treatment, giving a total area of 8.33 ha in 20 patches. To mimic realistic farm management practice, the locations of the patches within each treatment (Fig. 1) were selected by the farm manager. Each of the three ELS-X Treatment patches comprised a different seed mixture, referred to hereafter as Biennial, Bumblebird and Deluxe; the details of plant species and sowing rates are given in Table 1. Bumblebird and Deluxe were sown annually, but the kale *Brassica oleracea* and fodder beet *Beta vulgaris* in the Biennial mix produced seed in their second year, following seeding by triticale (wheat/rye hybrid) and quinoa *Chenopodium quinoa* in the first year. The ELS Treatment patches were all sown with the Biennial mixture. One of the ELS Treatment patches failed to establish and hence the sample sizes of the different mixtures were nine, five and five for Biennial, Bumblebird and Deluxe respectively. In 2007/08, the Biennial patches were in their second year, i.e. the kale and fodder beet were in seed.

Maximum seed production was estimated from samples taken from the patches during September each year. In each of the patches, five quadrats (1 m × 0.5 m) were sampled arbitrarily, avoiding the first 3 m from the edge, but utilising the whole patch. The number of individuals of all plant species (i.e. sown species and weeds rooted in a quadrat) were counted and recorded, along with their reproductive status (seeding/non-seeding). All seeds and seed heads from each of the seeding species were collected and stored separately in labelled plastic bags. If samples were not processed immediately they were frozen. When processed, the seed from each species was dried at 80ºC for 24 hours and then weighed.

Bird use of the bird food patches, and relative use of each patch type, was monitored throughout the winter in 2007/08. Counts of birds, identified to species, present in all 20 sown patches were made on each of six visits (October, November, December, early January, late January and February). To minimise the risk of multiple counts of birds moving between patches, all patches were counted on the same day at approximately the same time by five observers each counting four patches. Patches were observed at a distance and eventually flushed by walking around the patch perimeter and then through the patch. The aim of the counts was to record the total number of birds in each patch and counts took *c.* 20 minutes to one hour to complete depending on bird numbers and species composition and the vegetation density of the patch. To obtain a comparison with bird use of cropland, each patch was paired with an equivalent area of crop, in a similar shape and location, in an adjacent field, which was counted immediately after the patch count.

Late winter use of other patch and margin habitats

Three counts (not reported here) of the bird food patches made in the previous winter (i.e. 2006/07), and results from other studies (e.g. Siriwardena *et al.* 2008), indicated that bird usage declined in late winter (late January/February, see below). Thus potential bird use of alternative prescribed habitats, i.e. margins and other patch types, was also monitored in January to the beginning of April in 2008. Samples of different margin and patch types, including bird food patches, located chiefly in the ELS-X Treatments, were counted 12 times, the observer walking at a steady pace and recording all birds seen or flushed from each habitat type. The habitat types, and the total area of each sample, are given in Table 2. The numbers of birds encountered in all habitats except the bird food patches were small and thus results were expressed as the numbers of birds per habitat type, and numbers per unit area, totalled across all 12 visits.

Winter bird population census

The birds present in winter were also recorded at the scale of each treatment (i.e. CC, ELS and ELS-X) in each of the five replicate blocks. Birds were monitored using hedgerow transects located in the interior of each treatment (with one exception

where *c.* one third of the transect in one CC treatment comprised a treatment boundary; Fig. 1). Within each treatment, the hedges were usually contiguous, with a few instances of interspersed small copses, clumps of bushes/trees and gaps. Selected structural details of the 15 transects are given in the supplementary material (Table S1). All birds in the hedges, hedgerow trees and within 10 m of either side were recorded by spot mapping, bird locations and activities being recorded on large-scale maps which included the locations of all hedgerow trees and other landmarks. Birds in the transects were recorded on three visits (November, December and January) in each of the winters of 2005/06 and 2006/07, i.e. before and after bird food patch establishment. Counts began a little after dawn and finished at least two hours before dusk; counts were not undertaken in weather likely to depress bird activity, such as rain or high winds. Each count was usually completed within two days, occasionally three in the event of poor weather. The bird records were later digitised using the LiDAR canopy height model within a geographic information system (ArcMAP v9.3 © 2008 ESRI Inc.). The birds recorded in the winter comprised a mixture of residents, short and regional-scale migrants and long distance migrants in unknown proportions. Therefore, the overall winter abundance of each species was expressed as the number of individuals per kilometre per transect totalled across all three counts. Totals (per km) were also calculated for each transect for 'all species' and for 'granivorous species' (Chaffinch *Fringilla coelebs*, Linnet, Yellowhammer *Emberiza citrinella*, Reed Bunting *Emberiza schoeniclus*, Tree Sparrow *Passer montanus*).

Breeding bird territory census

The breeding birds within each treatment were recorded by territory mapping using the same 15 hedgerow transects (Table S1) used for the winter bird counts. Birds, and their activities, in the hedges and hedgerow trees and within 10 m either side were mapped as in the winter. Particular attention was paid to behaviour indicative of breeding. Four visits (April, May, June and early July) were made in the breeding seasons of 2006 and 2007, i.e. before and after the availability of the winter bird food patches. The bird records were again digitised and territory locations were estimated using observer judgement based on recorded bird behaviour to assign each record to a territory (Bibby *et al.* 1992), and then generating minimum convex polygons to

represent approximate territory boundaries. A 10-m buffer was placed around the resultant polygons, in order to account for possible error when census mapping and digitizing locations. Territories estimated to occur wholly or substantially (more than two thirds minimum convex polygon area) within each treatment were counted for each species. The total numbers of territories per kilometre per transect were then calculated for each individual species and also for the two groups of 'all species' and 'granivorous species'.

Data analysis

To account for non-normality and the non-negative integer property of the data, a Poisson modelling approach was used. Where the fit of Poisson models was poor, *P* values were corrected using the Quasipoisson adjustment, all models being run in R 2.10.0 (R Development Core Team 2009).

To examine differences in bird numbers (for all species combined and for individual species) between patch types for each visit, generalized linear models with a Poisson distribution for the bird counts and log-link function were used, with log (patch area) as an offset to allow for differences in patch area. Patch type (Biennial, Bumblebird, Deluxe) was used as a predictive factor plus length of woody boundary to accommodate any effects of patch boundary structure. Models fitting individual slopes for each patch type were compared to those fitting a common slope.

The relationships between individual patch seed production, measured in September (kg of seed per patch, all plant species) and the bird counts (all species combined) from October to February were examined using generalized linear models with a Poisson distribution for the bird counts, patch type as a factor plus seed yield and an interaction between type and seed yield. When the interaction was not significant, models using a single slope and different intercepts for patch type were compared with those fitting a single slope and intercept. Area was not included as an offset because the response being modelled was the observed response of the birds to the actual seed yield of each patch.

Differences in bird numbers both in the winter and during the breeding season in the years before and after bird food patches were available were compared at the whole farm scale using generalized linear models with a Poisson distribution for the

bird counts (all species combined, granivorous species and individual species) for each transect. The influence of Chaffinch, the most numerous species in both seasons, was also examined by removing numbers of Chaffinches from the totals for 'all species' and 'granivorous species'. Differences between years were assessed by comparing models including transect and year as factors with models omitting year effects. To examine treatment-level (CC, ELS, ELS-X) effects on differences between years for 'all species' and 'granivorous species', the models were re-run including treatment as a factor and an interaction between treatment and year. Models with and without the interaction were then compared. Where treatment had a significant effect, differences between years were examined separately for each treatment using transect and year as factors.

RESULTS

Winter bird food patches

As expected, the abundance and species richness of birds counted on the bird food patches were substantially higher than those recorded in equivalent areas of crop. In 2007/08 (Biennial mixture in its second year), the mean number of birds and the mean species richness per patch per visit (all patches combined) standardised to an area of 1 ha was 79 and 8.7 respectively compared with 0.3 birds and 0.2 species in 1 ha of crop. Considering PSA species only, these figures were 27 birds and 2 species in patches compared with 0.06 birds and 0.02 species in crops. The PSA species recorded in patches were Linnet, Goldfinch *Carduelis carduelis*, Greenfinch, Yellowhammer, Reed Bunting, Skylark *Alauda arvensis*, Grey Partridge *Perdix perdix* and Kestrel and those recorded in the crop were Skylark and Grey Partridge.

The numbers of birds counted varied between patch types (Fig. 2), but due to large variation in the numbers present on individual patches (see below), the only significant difference occurred in November ($F_{2,16}$ = 8.587, P = 0.003, Fig. 2). There was no effect of length of woody boundary in any month. Overall, the Bumblebird mixture attracted fewer birds and in general, bird numbers on all patch types declined in late winter (late January, February). There was large variation in the responses of individual species to the different patch types; examples are shown in Figure 3. Apart from an initial

presence in Deluxe patches in early winter, Yellowhammers were rare whereas Linnet numbers were maintained in Deluxe until late winter (Figs. 3a and 3b). Song Thrushes *Turdus philomelos* showed an increasing trend in the Biennial mix until late winter (Fig. 3c) and Dunnocks *Prunella modularis* occurred in all three patch types (Fig. 3d). Chaffinches (Fig. 3e) showed a similar pattern to that for all species combined (Fig. 2) while Greenfinches (Fig. 3f) showed a similar response to Yellowhammers to Deluxe and a peak in numbers in the other two types in December. Goldfinches (Fig. 3g) were commonest in the Biennial mix, declining in numbers here throughout the winter, but were, at least in part, attracted by chicory *Cichorium intybus* which had occurred as an accidental contaminant. Reed Buntings (Fig. 3h) occurred sporadically in low numbers, but were absent from the Biennial mix apart from a single bird in one patch in early January. As with the results for all species combined, the large variation in counts of individual species between individual patches caused most differences between patch types to be either non-significant or to occur sporadically in different months for different species (Table 3). The exceptions were the increasing and/or decreasing trends for Song Thrush, Chaffinch and Goldfinch (Figs. 3c, e and g). A significant boundary effect was found on three occasions (Yellowhammer, October, $F_{1,16}$ = 11.249, P = 0.004; Greenfinch, October, $F_{1,16}$ = 6.295, P = 0.024; Reed Bunting, October, $F_{1,16}$ = 10.588, P = 0.005) and implied a negative relationship between bird numbers and the length of woody boundary. However, for Yellowhammer and Reed Bunting, birds were only present in two and three patches respectively and the numbers of the latter species were low (1, 2 and 5 birds per patch). Therefore, these results should be treated with caution and additional data to elucidate possible boundary effects are currently being collected.

A positive relationship between the amount of seed produced by a patch and the total numbers of birds present at the beginning of the winter was expected. However, the relationship was significant in November only (slope = 0.00152, $F_{1,18}$ = 8.297, P = 0.010, Fig. 4), the best model using a single slope and intercept. There was no evidence of a relationship in either October (slope = -0.00030, $F_{1,16}$ = 0.095, P = 0.762) or December (slope = 0.00127, $F_{1,16}$ = 1.212, P = 0.288). This was probably due to the large variation in seed yield between patches, including within the same type, and also between bird counts on individual patches. Relationships between seed yields and bird

counts later in the winter were not expected (due to the length of time from measurement of the seed crop in September) and none were found (Early January: slope = -0.00044, $F_{1,16}$ = 0.090, P = 0.768; Late January: slope = -0.00013, $F_{1,16}$ = 0.016, P = 0.901; February: slope = -0.00075, $F_{1,16}$ = 0.354, P = 561).

Late winter use of other patch and margin habitats

The largest numbers of birds were recorded on the bird food patches (Table 2) with only small numbers recorded on any of the other margin or patch habitat types. The most frequent species on tussocky margins were Goldfinch and Reed Bunting, the former being attracted to teasels *Dipsacus fullonum*. In pollen and nectar margins, partridges (mostly Red-legged Partridge *Alectoris rufa*) and Chaffinch were most frequent. No species were associated with any of the other habitat types, and in all cases, the numbers of birds present was small (Table 2).

Winter bird population census

At a whole farm-scale, using the 15 transects as replicates, more birds were recorded in the winter (2006/07) following bird food patch establishment than in that (2005/06) before (Table 4). Numbers were greater post-establishment for all the species recorded and the differences were significant for the two groups of 'all species' and 'granivorous species' and for nine of the individual species (Table 4). The increase between years for 'all species' and 'granivorous species' remained significant when Chaffinch was excluded from the groups (all species: $F_{1,14}$ = 6.692, P < 0.001; granivorous species: $F_{1,14}$ = 6.810, P < 0.001). At the treatment-level, for 'all species' and 'granivorous species' numbers increased post-establishment in the form of ELS-X > ELS > CC (Fig. 5a), but treatment was significant only for 'granivorous species' ($F_{2,12}$ = 6.483, P = 0.012). For 'granivorous species', the difference between years was significant for all three treatments (CC: $F_{1,5}$ = 17.009, P = 0.015; ELS: $F_{1,5}$ = 36.161, P = 0.004; ELS-X: $F_{1,5}$ = 92.740, P < 0.001; Fig. 5a).

Breeding bird territory census

More breeding territories were recorded in the breeding season (2007) following bird food patch establishment than in that (2006) before (Table 5). Numbers of territories increased post-establishment (albeit marginally for some species) for all the species recorded except Reed Bunting (resident species) and Whitethroat (summer migrant) and the differences were significant for 'all species', 'granivorous species', Chaffinch, Dunnock and Robin, and nearly so (P = 0.053) for Linnet (Table 5). However, the significance for 'granivorous species' was due to Chaffinch (without Chaffinch: $F_{1,14}$ = 2.354, P = 0.125), but that for 'all species' was independent of Chaffinch numbers (without Chaffinch: $F_{1,14}$ = 5.390, P = 0.020). There were no significant differences at the Treatment-level; unlike the winter data, the pattern of numbers across the three treatments was similar in both years for both groups of species (Fig. 5b).

DISCUSSION

It was no surprise that many more birds were counted on the bird food patches than on equivalent areas of crop (e.g. Boatman *et al.* 2000, Henderson *et al.* 2004, Stoate *et al.* 2004, Field *et al.* 2009), but the winter census based on the 15 transects indicated an increase in bird numbers across the farm in general, not just on the patches. This implied that birds do not simply move between food-rich patches across the landscape, but also use intervening habitat of different types. A response to habitat other than, or in addition to, the bird food patches was also indicated by the results for Yellowhammer (Whittingham *et al.* 2005) and Blue Tit. Both these species showed a significant increase in numbers in the winter of 2006/07 (Table 4), but neither was present in large numbers on the bird food patches (Fig. 3). Overall, the species showing the greatest response to the patches in winter were, as expected, the seed-eaters, plus Song Thrush and Dunnock (see below). With the exception of Blue Tit (which appears to be less sensitive to habitat quality than Great Tit, Hinsley *et al.* 1999), woodland or woodland edge species such as Robin and Wren tended to show little response.

The increase in the numbers of breeding territories in 2007, following patch establishment, also suggested that the birds were responding at a whole farm-scale and that, given the availability of suitable habitat, had been encouraged to remain in the area to breed (Gillings *et al.* 2005). The fact that most of the trends in both winter

and breeding season bird numbers were positive (Tables 4 & 5) suggested a positive effect of the overall habitat provision under ES. The general increase in the numbers of breeding territories at Hillesden from 2006 to 2007 contrasted with the changes in the national English population monitored by the BTO/JNCC/RSPB Breeding Bird Survey (Risely *et al.* 2008, Table 5). At the national scale, six species showed little change or an increasing trend from 2006 to 2007, whereas seven, including Linnet and Yellowhammer, showed a declining trend. This, together with the fact that there was no other major habitat alteration on the farm between the two winters and that the bulk of the census work was carried by the same two observers in all years, suggested that there was a genuine positive response to the ES habitat management.

Bird exploitation of dedicated food patches will depend on many factors including seed type and yield, patch location, and the distribution and abundance of alternative food supplies in the wider landscape (Whittingham & Evans 2004, Siriwardena & Stevens 2004, Stoate *et al.* 2004, Siriwardena *et al.* 2006, Siriwardena 2010, in press, this issue). Some species, such as Dunnock and Song Thrush, may also respond to attributes other than those directly related to the sown seed crop such as shelter, access to damp ground and associated weed seed and invertebrate food resources (Peach *et al.* 2004, Gilroy *et al.* 2008). This was thought especially pertinent for Song Thrush (and other thrushes) in the tall dense cover offered by the Biennial patches (Fig. 3, Table 3). Similarly, the presence of Linnets in the Deluxe patches (Fig. 3b) was thought to be due in part to growth of weeds (e.g. chickweed *Stellaria media* and groundsel *Senecio vulgaris*) and, as mentioned above, Goldfinches responded to the accidental presence of chicory in the Biennial mix (Fig. 3g, Table 3). The range of responses of individual species to the different seed mixtures used at Hillesden indicated that selecting a single 'best' mix was unlikely to be feasible; different bird species are well known to prefer different seed types and sizes (Wilson *et al.* 1999, Boatman & Stoate 2002, Stoate *et al.* 2004, Holland *et al.* 2006). There is also the choice of annual versus biennial mixes; the results for the Biennial mix reported here were dominated by the characteristics of the second year-seeding species, kale and fodder beet. Therefore, provision of a range of patch types, within a single farm or co-ordinated across larger areas, could be most cost-effective.

There is also the possibility that particular bird species could be targeted in particular locations. Such an approach is being trialled at a regional scale (Phillips 2009), but there is no reason why targeting could not take place at a farm or even field scale to benefit certain local populations, e.g. a colony of Tree Sparrows. In addition, the physical attributes of the patches differed substantially, e.g. in terms of sown species heights and densities, and this variation was increased enormously by additional variation in growth performance and seed production of individual patches (Fig. 4). Bird food patches, and other non-crop habitat types, tend to be located in the least favourable/least productive areas of fields, and this, coupled with the repeat sowing of the same areas across several years can result in poor establishment and performance. Thus the quality of ES options is likely to be as important as their identity; the production and maintenance of good quality bird food, and other semi-natural habitat will probably require a similar management effort to that devoted to crops (Stoate *et al.* 2004, Siriwardena & Anderson 2007, Douglas *et al.* 2009, Lobley *et al.* 2009).

Although a positive relationship between increasing seed yield and bird numbers was expected, this was found only in November probably due to the large variation in both variables (Fig. 4). Many factors could have contributed to low bird counts such as random disturbance of a patch, e.g. by a Sparrowhawk *Accipiter nisus*, and 'normal' movements of flocks between patches and other habitat. Ideally, patches could have been counted more frequently or observed for extended periods, but the result for November did indicate that more productive patches are likely to attract more birds, at least in early winter. There was an indication that the nature of the boundary of a patch might influence bird numbers, and it is well known that certain species prefer open landscapes and avoid otherwise suitable habitat in close proximity to woodland and tall hedges (Chamberlain *et al*. 2009). The patches at Hillesden are bordered in part by woody vegetation sometimes including lines of trees. It is possible that this degree of shelter deters some species, but equally it might attract others. The important point is that location can influence the value of ES habitat to birds (Siriwardena & Stevens 2004) and thus patches with persistently low usage are probably best relocated elsewhere. We are currently collecting more data on bird use

of patches in relation to their location and boundary features to investigate species-specific responses to these factors.

Despite the large variation in the use of the three seed mix types (Figs. 2, 3 and 4), they all showed a sharp drop in bird numbers in late January and early February, the beginning of the so-called 'hungry gap' (Siriwardena *et al.* 2008) when both managed and natural food supplies become depleted (Hinsley *et al*. 2010) and/or less accessible to some species due to seed drop. Filling this gap may require novel seed crops or greater areas of dedicated bird food and there is also the possibility of 'artificial' feeding. However, the latter often involves the use of cereals and such relatively large and hard seed may not be suitable for all species. Even though sown crops may be depleted in late winter, providing these additional/alternative resources earlier in the winter may prolong the availability of 'natural', i.e. non-cropland, food supplies. The other types of ES habitat patches and margins were little used by birds in late winter, but this was perhaps not surprising given that they are not designed to provide winter foraging habitat for birds. It does however strengthen the case for promoting winter bird food patches as a preferred ES option. It might also be possible to increase the value of other types of ES habitat for birds in winter by, for example, allowing grass margins to go to seed (Buckingham & Peach 2006) and including additional plant species such as teasel. There is also the question of where the birds go in late winter. Our observations suggest that they move more widely in the landscape than the farm-scale and this is consistent with various studies of winter food use (Robinson *et al.* 2004, Siriwardena *et al.* 2006) and of winter bird movements in general (e.g. Prŷs-Jones 2002). This in turn suggests that consideration of local landscape factors and co-ordination of ES management at greater than a farm-scale will be necessary to maximise effectiveness for farmland birds. The problems of effective food provision in late winter in relation to bird movements in the landscape are discussed in detail by Siriwardena (2010, in press, this issue).

A range of possible reasons for the continuing decline in the FBI, despite the substantial investment in ES (and other agri-environment schemes), have been suggested. These include a lack of uptake of key options, problems with the quality, location and management of options and the possibilities of ES benefits being off-set by other (unknown) factors or being obscured by time-lags in the bird population

response and/or the detectability of such a response at a national scale (Chamberlain *et al.* 2000, Risely *et al.* 2008, P.V. Grice, *pers. comm*.). Option choice, quality, location and management have been discussed briefly above. At the farm-scale of the Hillesden experiment, there was no time lag in the response of bird numbers in winter to the presence of bird food patches, but the response in terms of numbers of breeding territories, although encouraging, was less clear-cut. Although the overall numbers of territories increased, the responses of individual species were more varied, and often non-significant (Table 5). Despite the contrast with the English national population trends for 2006 to 2007 (Table 5), a single year's results may reflect local population redistribution rather than a general increase; detecting a long-term, sustained effect will be influenced by many events, not least the recent cold winters.

Given the caveat of the current short-term nature of the monitoring at Hillesden, and the fact that, although it is a farm-scale experiment, it is a single site, the results suggest that winter survival is a key requirement that can be addressed by providing food patches. In the second winter, there was a clear trend for bird numbers to be higher in the ELS and ELS-X Treatment areas (Fig. 5a), much as expected from the locations of the patches. With three patches, the resources in the ELS-X Treatment were greater than those of standard ELS, but the latter could be increased by simply increasing patch size. In spring, the pattern of the numbers of territories across the treatments was largely similar both before and after the patches were established; Fig. 5b), suggesting that territory distribution occurred on a larger scale than that of the individual treatment blocks. Thus winter survival appeared to be key (but see Siriwardena *et al.* 2000), but sufficient potential breeding habitat to support an increase in the breeding population (Siriwardena & Anderson 2007) was also implicit in the results. Smaller-scale effects, of the other ES margin and habitat types, as well as the semi-natural (hedgerow, tree, woodland, riverine etc.) and anthropogenic (gardens, farmyards etc.) habitats might be expected to influence the birds at the scale of individual territories. Other taxa within the experiment, e.g. solitary bees and some butterflies, could also be expected to respond to habitat on a smaller scale than that of the birds. Future analyses will use the remote sensed habitat data to investigate such smaller-scale effects, and in particular, the breeding productivity of individual bird territories.

In conclusion, the provision of winter food increased the numbers of birds present at a farm and sub-farm scale in winter and there was some evidence that this, perhaps coupled with the availability of other ES habitats (Table 2), subsequently increased the breeding population. However, settlement to breed requires that sufficient nesting/foraging habitat is available to support an increase. Thus the management of ES as a scheme must ensure the uptake of a suite of options that provides both winter food supplies (productive patches in accessible locations) as well as spring/summer nesting and foraging habitat.

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REFERENCES

Anon. 2009. *Entry Level Stewardship Handbook (3rd edition).* London: Natural England.

Bibby, C.J., Burgess, N.D. & Hill, D.A. 1992. *Bird Census Techniques*. London: Academic Press.

Boatman, N.D., Stoate, C. & Watts, P.N. 2000. Practical management solutions for birds on lowland arable farmland. In Aebischer, N.J., Evans, A.D., Grice, P.V. &

Vickery, J.A. (eds) *Ecology and Conservation of Lowland Farmland Birds*: 105-114. Tring: British Ornithologists' Union.

Boatman, N.D. & Stoate, C. 2002. Growing crops to provide food for seed-eating farmland birds in winter. *Aspects Appl. Biol*. 67: 229-236.

Bradbury, R.B., Browne, S.J., Stevens, D.K. & Aebischer, N.J. 2004. Five-year evaluation of the impact of the Arable Stewardship Pilot Scheme on birds. *Ibis* 146 (Suppl. 2): 171-180.

Buckingham, D.L. & Peach, W.J. 2006. Leaving final-cut grass sileage in situ overwinter as a seed resource for declining farmland birds. *Biodiv. Conserv*. 15: 3827-3845.

Carvell, C., Rothery, P., Pywell, R.F. & Heard, M.S. 2008. Effects of resource availability and social parasite invasion on field colonies of *Bombus terrestris*. *Ecol. Entomol*. 33: 321-327.

Chamberlain, D.E., Fuller, R.J., Bunce, R.G.H., Duckworth, J.C. & Shrubb, M. 2000. Changes in the abundance of farmland birds in relation to the timing of agricultural intensification in England and Wales. *J. Appl. Ecol*. 37: 771-788.

Chamberlain, D., Gough, S., Anderson, G., Macdonald, M., Grice, P. & Vickery, J. 2009. Bird use of cultivated fallow 'Lapwing plots' within English agri-environment schemes. *Bird Study* 56: 289-297.

Davey, C.M., Vickery, J.A., Boatman, N.D., Chamberlain, D.E., Parry, H.R. & Siriwardena, G.M. 2010. Assessing the impact of Entry Level Stewardship on lowland farmland birds in England. *Ibis*. In press.

Defra. 2008. Wildbird populations 2007. *Sustainable development strategy and public service agreement indicators*, pp. 22. London: Environment Statistics Service, Defra.

Defra. 2009a. http://www.defra.gov.uk/environment/statistics/wildlife/kf/wdkf03.htm, accessed May 2009.

Defra 2009b. https://statistics.defra.gov.uk/esg/ace/a3_data.htm, accessed May 2009.

Donald, P.F. & Evans, A.D. 2006. Habitat connectivity and matrix restoration: the wider implications of agri-environment schemes. *J. Appl. Ecol*. 43: 209-218.

Douglas, D.J.T., Vickery, J.A. & Benton, T.G. 2009. Improving the value of field margins as foraging habitat for farmland birds. *J. Appl. Ecol*. 46: 353-362.

Field, R.H., Morris, A.J., Grice, P.V. & Cooke, A. 2009. Winter use of seed-bearing crops by birds within the English Environmental Stewardship scheme. Abstract only. http://bouproc.blogspot.com/2010/06/lowland-farmland-birds-3-abstract-field.html

Fuller, R.J. 2000. Relationships between recent changes in lowland British agriculture and farmland bird populations: an overview. In Aebischer, N.J., Evans, A.D., Grice, P.V. & Vickery, J.A. (eds) *Ecology and Conservation of Lowland Farmland Birds*: 5-16. Tring: British Ornithologists' Union.

Fuller, R.J., Gregory, R.D., Gibbons, D.W., Marchant, J.H., Wilson, J.D., Baillie, S.R. & Carter, N. 1995. Population declines and range contractions among lowland farmland birds in Britain. *Conserv. Biol*. 9: 1425-1441.

Gillings, S., Newson, S.E., Noble, D.G. & Vickery, J.A. 2005. Winter availability of cereal stubbles attracts declining farmland birds and positively influences breeding population trends. *Proc. Roy. Soc. Lond. B*. 272: 733-739.

Gilroy, J.J., Anderson, G.Q.A., Grice, P.V., Vickery, J.A., Bray, I., Watts, P.N. & Sutherland, W.J. 2008. Could soil degradation contribute to farmland bird declines? Links between soil penetrability and the abundance of Yellow Wagtails *Motacilla flava* in arable fields. *Biol. Conserv*. 141: 3116-3126.

Gregory, R.D., Noble, D.G. & Custance, J. 2004. The state of play of farmland birds: population trends and conservation status of lowland farmland birds in the United Kingdom. *Ibis* 146 (Supple. 2): 1-13.

Gruar, D., Barritt, D. & Peach, W.J. 2006. Summer utilization of oilseed rape by Reed Buntings *Emberiza schoeniclus* and other farmland birds. *Bird Study* 53: 47–54.

Henderson, I.G., Vickery, J.A. & Carter, N. 2004. The use of winter bird crops by farmland birds in lowland England. *Biol. Conserv.* 118: 21-32.

Hill, R.A. & Thomson, A.G. 2005. Mapping woodland species composition and structure using airborne spectral and LiDAR data. *Int. J. Remote Sens.* 26: 3763-3779.

Hill, R.A. & Broughton, R.K. 2009. Mapping understorey from leaf-on and leaf-off airborne LiDAR data of deciduous woodland. *ISPRS J. Photogramm. Remote Sens.* 64: 223-233.

Hinsley, S.A., Rothery, P. & Bellamy, P.E. 1999. Influence of woodland area on breeding success in Great Tits (*Parus major*) and Blue Tits (*Parus caeruleus*). *J. Avian Biol.* 30: 271-281.

Hinsley, S.A., Hill, R.A., Gaveau, D.L.A. & Bellamy, P.E. 2002. Quantifying woodland structure and habitat quality for birds using airborne laser scanning. *Funct. Ecol.* 16: 851-857.

Hinsley, S.A., Hill, R.A., Bellamy, P.E., Harrison, N.M., Speakman, J.R., Wilson, A.K. & Ferns, P.N. 2008. Effects of structural and functional habitat gaps on breeding woodland birds: working harder for less. *Land. Ecol.* 23: 615-626.

Hinsley, S.A., Novakowski, M., Heard, M., Bellamy, P.E., Broughton, R.K., Hulmes, S., Hulmes, L., Peyton, J. & Pywell, R.F. 2010. Performance and effectiveness of winter bird food patches established under Environmental Stewardship: Results from the Hillesden experiment. *Aspects Appl. Biol*., in press.

Hole, D.G., Whittingham, M.J., Bradbury, R.B., Anderson, G.Q.A., Lee, P.L.M., Wilson, J.D. & Krebs, J.R. 2002. Widespread local House Sparrow extinctions – Agricultural intensification is blamed for the plummeting populations of these birds. *Nature, Lond*. 418: 931-932.

Holland, J.M., Hutchison, M.A.S., Smith, B. & Aebischer, N.J. 2006. A review of invertebrates and seed-bearing plants as food for farmland birds in Europe. *Annals Appl. Biol.* 148: 49–71.

Lobley, M., Saratsi, E. & Winter, M. 2009. Training and advice for agri-environmental management. http://www.bou.org.uk/bouproc-net/lfb3/lobley-etal.pdf

Moorcroft, D., Whittingham, M.J., Bradbury, R.B. & Wilson, J.D. 2002. The selection of stubble fields by wintering granivorous birds reflects vegetation cover and food abundance. *J. Appl. Ecol.* 39: 535–47.

Natural England. 2009. http://www.naturalengland.org.uk/ourwork/farming/funding/es/default.aspx, accessed May 2009.

Newton, I. 2004. The recent declines of farmland bird populations in Britain. An appraisal of causal factors and conservation actions. *Ibis* 146 (Suppl. 2): 579-600.

Pain, D.J. & Pienkowski, M.W. (eds) 1997. *Farming and Birds in Europe: The Common Agricultural Policy and its Implications for Bird Conservation*. London: Academic Press.

Peach, W.J., Denny, M., Cotton, P.A., Hill, I.F., Gruar, D., Barritt, D., Impey, A. & Mallord, J. 2004. Habitat selection by Song Thrushes in stable and declining farmland populations. *J. Appl. Ecol*. 41: 275-293.

Phillips, J., Winspear, R., Fisher, S. & Noble, D. 2009. Targeting agri-environment schemes for farmland birds in England. http://www.bou.org.uk/bouproc-net/lfb3/phillips-etal.pdf

Prŷs-Jones, R. 2002. Reed Bunting, *Emberiza schoeniclus*. In Wernham, C.V., Toms, M.P., Marchant, J.H., Clark, J.A., Siriwardena, G.M. & Baille, S.R. (eds) *The Migration Atlas: movements of the birds of Britain and Ireland*: 675-677. London: T. & A.D. Poyser.

R Development Core Team (2009). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, URL http://www.R-project.org.

Risely, K., Noble, D.G. & Baillie, S.R. 2008. *The Breeding Bird Survey 2007.* BTO Research Report 508. Thetford: British Trust for Ornithology.

Robinson, R.A. & Sutherland, W.J. 2002. Post-war changes in arable farming and biodiversity in Great Britain. *J. Appl. Ecol*. 39: 157-176.

Robinson, R.A., Hart, J.D., Holland, J.M. & Parrott, D. 2004. Habitat use by seed-eating birds: a scale-dependent approach. *Ibis* 146 (Suppl. 2): 87-98.

Schifferli, L. 2000. Changes in agriculture and the status of birds breeding in European farmland. In Aebischer, N.J., Evans, A.D., Grice, P.V. & Vickery, J.A. (eds) *Ecology and Conservation of Lowland Farmland Birds*: 17-25. Tring: British Ornithologists' Union.

Siriwardena, G.M. 2010. The importance of spatial and temporal scale for agri-environment scheme delivery. *Ibis*, in press.

Siriwardena, G.M., Baillie, S.R., Crick, H.Q.P., Wilson, J.D. & Gates, S. 2000. The demography of lowland farmland birds. In Aebischer, N.J., Evans, A.D., Grice, P.V. & Vickery, J.A. (eds) *Ecology and Conservation of Lowland Farmland Birds*: 117-133. Tring: British Ornithologists' Union.

Siriwardena, G.M. & Stevens, D.K. 2004. Effects of habitat on the use of supplementary food by farmland birds in winter. *Ibis* 146 (Suppl. 2): 144-154.

Siriwardena, G.M., Calbrade, N.A., Vickery, J.A. & Sutherland, W.J. 2006. The effect of the spatial distribution of winter seed food resources on their use by farmland birds. *J. Appl. Ecol*. 43: 628-639.

Siriwardena, G.M. & Anderson, G.Q.A. 2007. How can agri-environment measures providing winter food for birds best deliver population increases? *Aspects Appl. Biol.* 81: 117-126.

Siriwardena, G.M., Calbrade, N.A. & Vickery, J.A. 2008. Farmland birds and late winter food: does seed supply fail to meet demand? *Ibis* 150: 585-595.

Stoate, C., Henderson, I.G. & Parish, D.M.B. 2004. Development of an agri-environment scheme option: seed bearing crops for farmland birds. *Ibis* 146 (Supple. 2): 203-209.

Swash, A.R.H., Grice, P.V. & Smallshire, D. 2000. The contribution of the UK Biodiversity Action Plan and agri-environment schemes to the conservation of farmland birds in England. In Aebischer, N.J., Evans, A.D., Grice, P.V. & Vickery, J.A. (eds) *Ecology and Conservation of Lowland Farmland Birds*: 36-42. Tring: British Ornithologists' Union.

Vickery, J.A., Evans, A.D., Grice, P.V., Aebischer, N.J. & Brand-Hardy, R. (eds) 2004. *Ecology and Conservation of Lowland Farmland Birds II: The Road to Recovery*. *Ibis* 146 (Supple. 2).

Voříšek, P., Škorpilová, J., Klvaňová, A., Strien, A. van & Gregory, R.D. 2009. Trends in abundance and biomass of widespread European farmland birds: how much have we lost? http://www.bou.org.uk/bouproc-net/lfb3/vorisek-etal.pdf

Whittingham, M.J. & Evans, K.L. 2004. The effects of habitat structure on predation risk of birds in agricultural landscapes. *Ibis* 146 (Supple. 2): 210-220.

Whittingham, M.J., Swetnam, R.D., Wilson, J.D., Chamberlain, D.E. & Freckleton, R.P. 2005. Habitat selection by Yellowhammers *Emberiza citrinella* on lowland farmland at two spatial scales: implications for conservation management. *J. Appl. Ecol*. 42: 270-280.

Wilson, J.D., Morris, A.J., Arroyo, B.E., Clark, S.C. & Bradbury, R.B. 1999. A review of the abundance of invertebrate and plant foods of granivorous birds in northern Europe in relation to agricultural change. *Agri. Ecosys. Environ*. 75: 13–30.

Table 1. Composition at sowing of winter bird food patches. The Biennial mix (ELS and ELS-X) was sown at 40 kg ha^{-1}, the Deluxe (ELS-X) and Bumblebird (ELS-X) at 20 kg ha^{-1}.

%	Bumblebird	%	Deluxe	%	Biennial
40	Triticale (wheat/rye hybrid)	40	Triticale	70	Triticale
15	Millet (*Echinochloa esculenta*)	20	Millet	14	Kale (*Brassica oleracea*)
15	Dwarf Sunflower (*Helianthus annuus*)	20	Buckwheat (*Fagopyrum esculentum*)	14	Quinoa (*Chenopodium quinoa*)
10	Quinoa	10	Quinoa	2	Fodder Beet (*Beta vulgaris*)
10	Fodder Radish (*Raphanus sativus*)	10	Fodder Radish		
7	Sweet Clover (*Melliotus officinalis*)				
3	Borage (*Borago officinalis*)				

Table 2. Extent and use of different patch and margin habitat types in late winter (January to first week of April, 2008). EF1 (management of field corners: creation of grass and wildflower patches, 0.5 ha, in field corners that are awkward to reach with machinery) and EF2 (wild bird seed mixture patches: mixtures of annual and biennial small seed-bearing crop species sown in low yielding or awkward patches, 0.25-0.5 ha) are ES habitat designations (Anon 2009).

Habitat type	Total area recorded, ha	Total no. of birds counted	Birds per 100 m^2 All species	Granivorous species
Bird food EF2 patch	4.84	1144	2.36	1.81
Tussocky margin	1.34	121	0.90	0.72
Pollen & nectar margin	2.29	61	0.27	0.11
Flower EF1 patch	3.64	45	0.12	0.04
Annual cultivation margin	1.28	10	0.08	0.02
Natural regeneration margin	0.70	9	0.13	0.00

Table 3. Differences (illustrated in Figure 3) in the use of the three types of bird food patch by individual bird species during the winter of 2007/08. Results are shown for significance levels up to $P = 0.10$ (plus one additional result for Chaffinch); for all other results, P exceeded 0.10. Note that results for Reed Bunting should be treated with caution due to small numbers of birds (see Fig. 3h).

Species		Oct.	Nov.	Dec.	Early Jan.	Late Jan.	Feb.
Song Thrush	$F_{2,16}$	4.360	7.535	2.930	3.241	23.105	2.927
	P	0.031	0.005	0.082	0.066	< 0.001	0.083
Chaffinch	$F_{2,16}$	6.177	14.136	4.810	2.819	2.338	3.069
	P	0.010	< 0.001	0.023	0.089	0.129	0.074
Goldfinch	$F_{2,16}$	6.310	27.325	2.570	-	-	-
	P	0.010	< 0.001	0.108	-	-	-
Reed Bunting	$F_{2,16}$	2.452	3.115	5.354	-	-	-
	P	0.118	0.072	0.017	-	-	-
Dunnock	$F_{2,16}$	2.758	-	-	-	10.106	9.209
	P	0.093	-	-	-	0.001	0.002
Yellowhammer	$F_{2,16}$	5.806	-	-	-	-	-
	P	0.013	-	-	-	-	-

Greenfinch	$F_{2,16}$	3.790	-	-	-	-	-
	P	0.045	-	-	-	-	-

Table 3 cont.

Species		Oct.	Nov.	Dec.	Early Jan.	Late Jan.	Feb.
Linnet	$F_{2,16}$	-	-	-	-	2.718	-
	P	-	-	-	-	0.096	-

Table 4. Winter bird abundance (mean numbers of birds per kilometre of transect, *n* = 15) at a farm-scale before (2005/06) and after (2006/07) the establishment of winter bird food patches and other habitats.

	Winter			
Species	2005/06	2006/07	$F_{1,14}$	*P* value
All species	52.4	118.5	5.701	< 0.001
Granivorous species	9.9	51.0	3.370	< 0.001
Blackbird *Turdus merula*	10.0	10.8	0.068	0.798
Chaffinch *Fringilla coelebs*	8.4	27.7	4.336	< 0.001
Robin *Erithacus rubecula*	7.7	8.2	0.105	0.751
Blue Tit *Cyanistes caeruleus*	7.3	13.1	14.022	0.002
Song Thrush *Turdus philomelos*	4.6	8.0	5.857	0.030
Dunnock *Prunella modularis*	4.4	13.2	45.022	< 0.001
Wren *Troglodytes troglodytes*	4.4	5.6	1.516	0.238
Great Tit *Parus major*	2.7	5.3	9.002	0.010
Yellowhammer *Emberiza citrinella*	1.3	11.8	66.799	< 0.001

Linnet *Carduelis cannabina*	0.2	7.4	14.844	0.002
Tree Sparrow *Passer montanus*	0.1	0.7	28.186	< 0.001
Reed Bunting *Emberiza schoeniclus*	0.0	3.4	4.518	< 0.001

Table 5. Breeding season bird abundance (mean numbers of territories per kilometre of transect, *n* = 15) at a farm-scale before (2006) and after (2007) the establishment of winter bird food patches and other habitats. Results are compared with the % change from 2006 to 2007 in national populations (Risely *et al.* 2008). * = significant change.

	Breeding season				National trends (%)	
Species	2006	2007	$F_{1,14}$	*P* value	UK	England
All species	33.9	39.9	8.176	0.004	-	-
Granivorous spp.	13.0	16.0	5.307	0.021	-	-
Chaffinch	7.3	9.1	9.935	0.007	-2	-3
Yellowhammer	3.8	4.1	1.848	0.196	-5	-9
Dunnock	3.7	4.6	5.251	0.038	0	+1
Blackbird	3.0	3.2	0.516	0.485	4	4
Blue Tit	2.8	3.6	2.945	0.108	-6	-7
Robin	2.8	3.5	4.774	0.046	+3	+2
Great Tit	2.3	2.6	0.653	0.433	0	-5
Wren		2.1	2.2	0.059	0.808	+13* +12

Reed Bunting	1.2	1.1	0.028	0.869	-2	+1
Linnet	0.7	1.6	4.453	0.053	-3	-13
Song Thrush	0.6	0.7	0.581	0.459	+1	+1

Fig. 1. Map of the study area showing arrangement of the replicate blocks (B1 – B5) and the three treatments (CC, ELS and ELS-X) within each block. Blue lines show the locations of the 15 hedgerow transects and the black dots the 20 bird food patches. The insert shows the location of the study area in central lowland England.

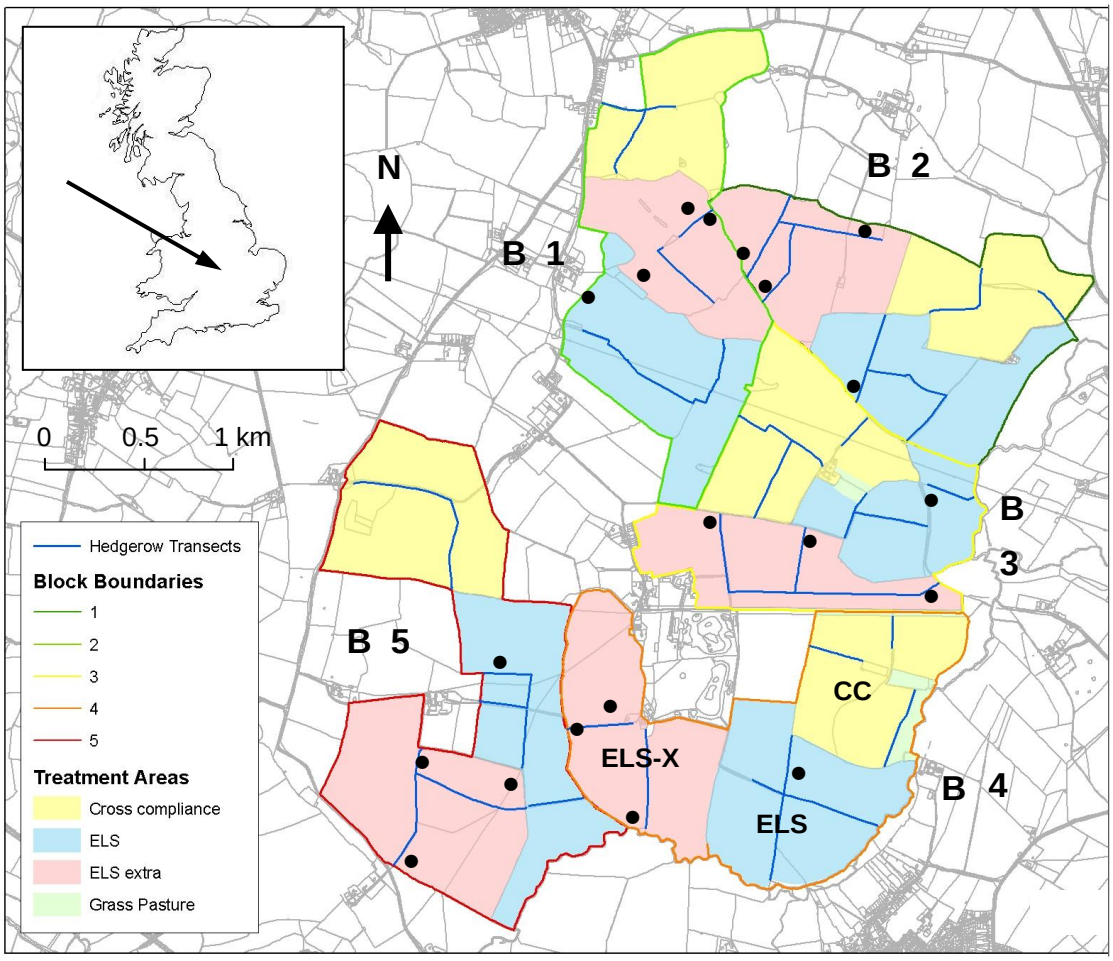


Fig. 2. Bird usage of the three different types of bird food patch during the winter of 2007/08. Bird numbers (for all species combined) are mean counts per visit date per patch type, corrected to a standard patch area of 0.5 ha. Date 0 = 1st October; error bars show + 1 se only for clarity.

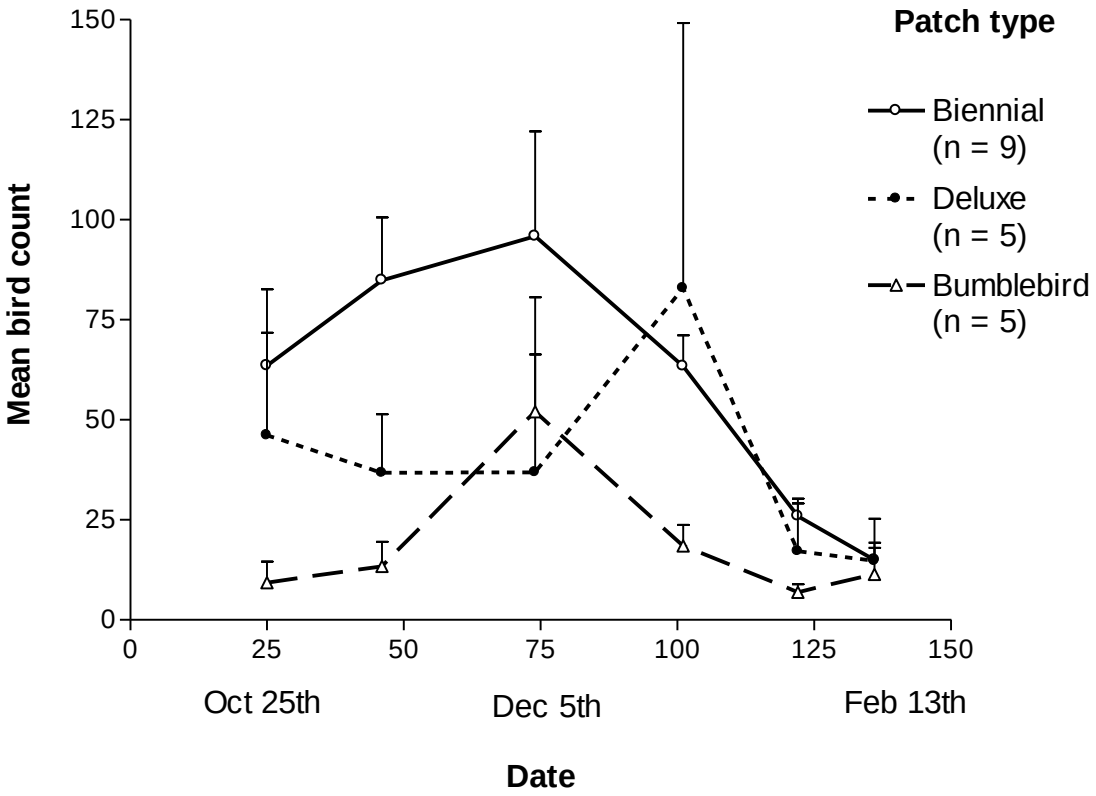


Fig. 3. Examples of individual bird species usage of the three different types of bird food patch during the winter of 2007/08. Bird numbers are mean counts per visit date per patch type, corrected to a standard patch area of 0.5 ha. Date 0 = 1st October; error bars show + 1 se only for clarity.

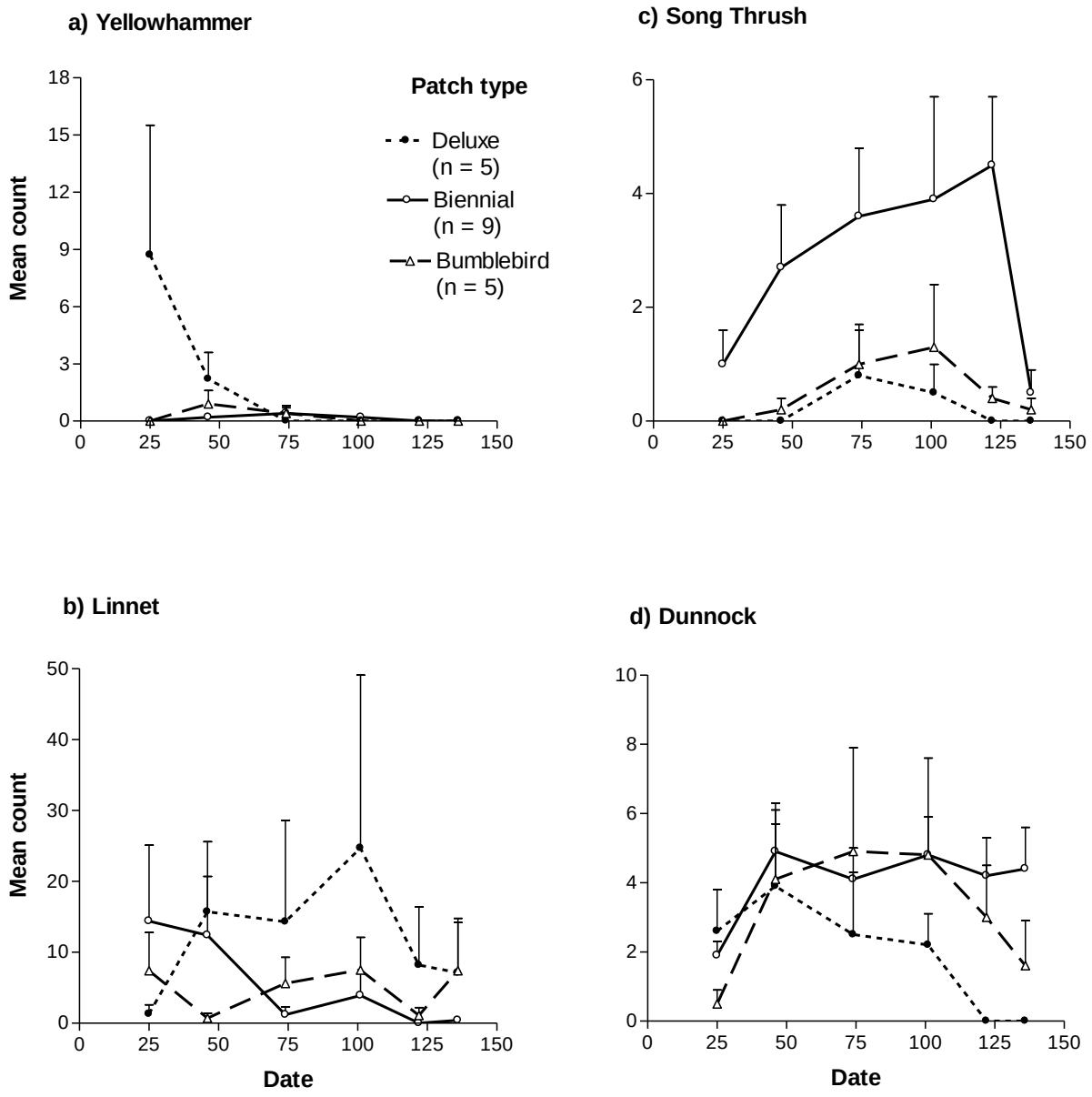


Fig. 3 cont.

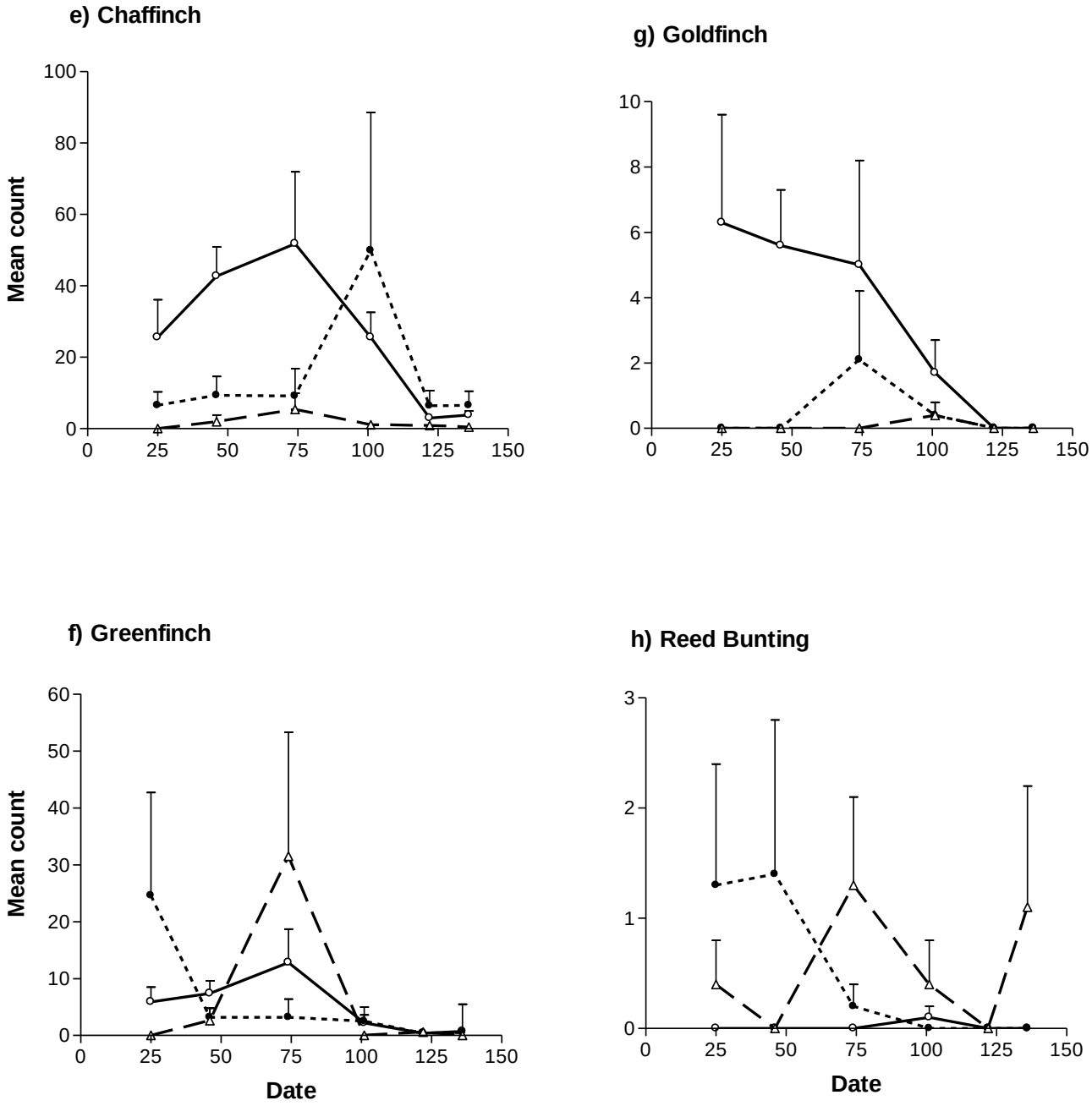

e) Chaffinch
g) Goldfinch
f) Greenfinch
h) Reed Bunting
Mean count
Date
0
25
50
75
100
125
150
20
40
60
80
2
4
6
8
10
30
1
3

Fig. 4. Effect of seed yield on bird usage of individual patches in November 2007. Fitted line calculated from Poisson model using single slope and intercept. (Note that the results include data for the failed Biennial mix patch which was resown in 2007 with the Deluxe annual mix, and that no seed crop data were available for one of the Bumblebird patches.)

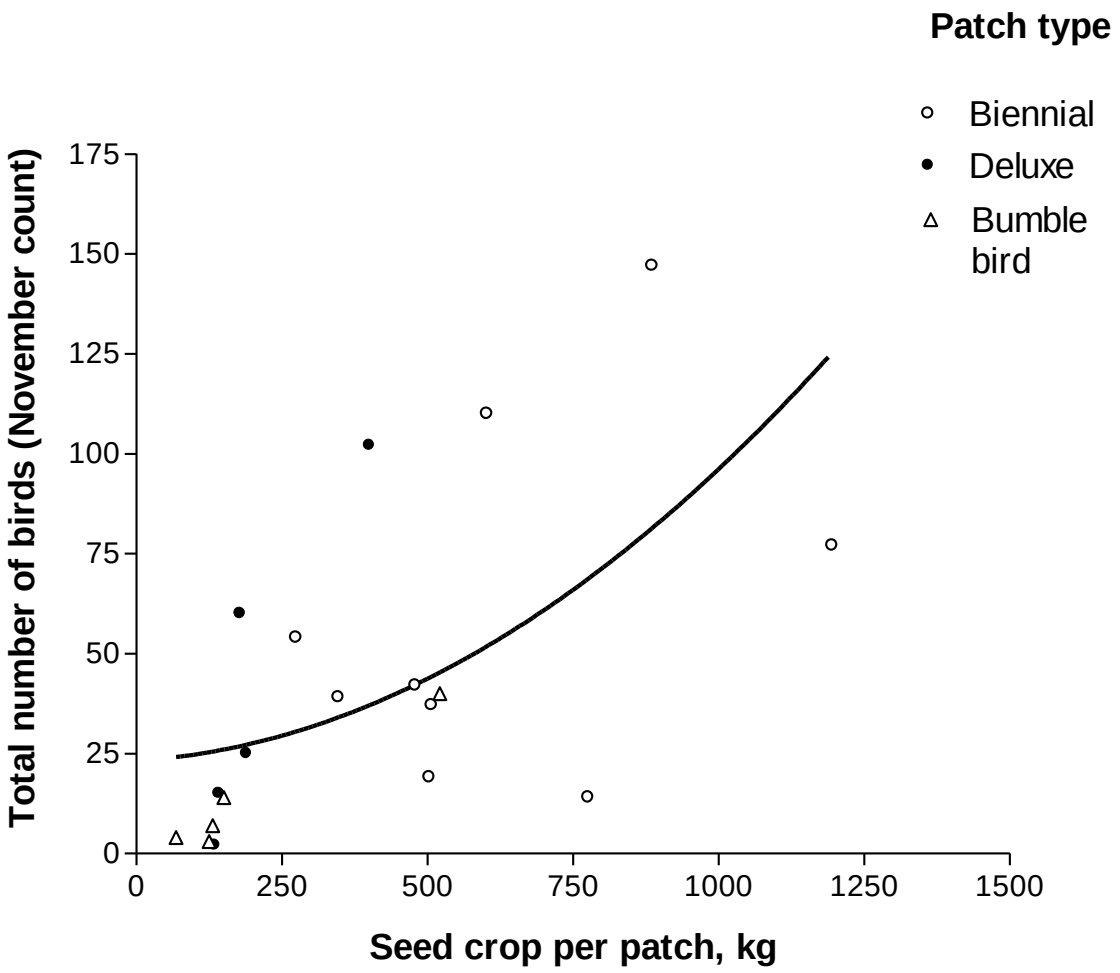
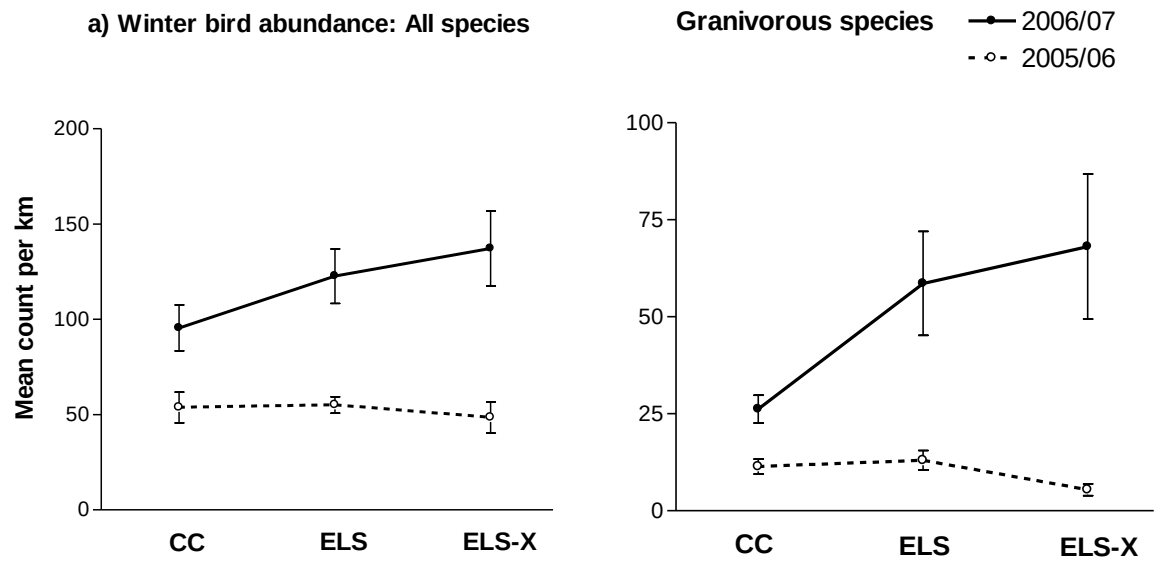
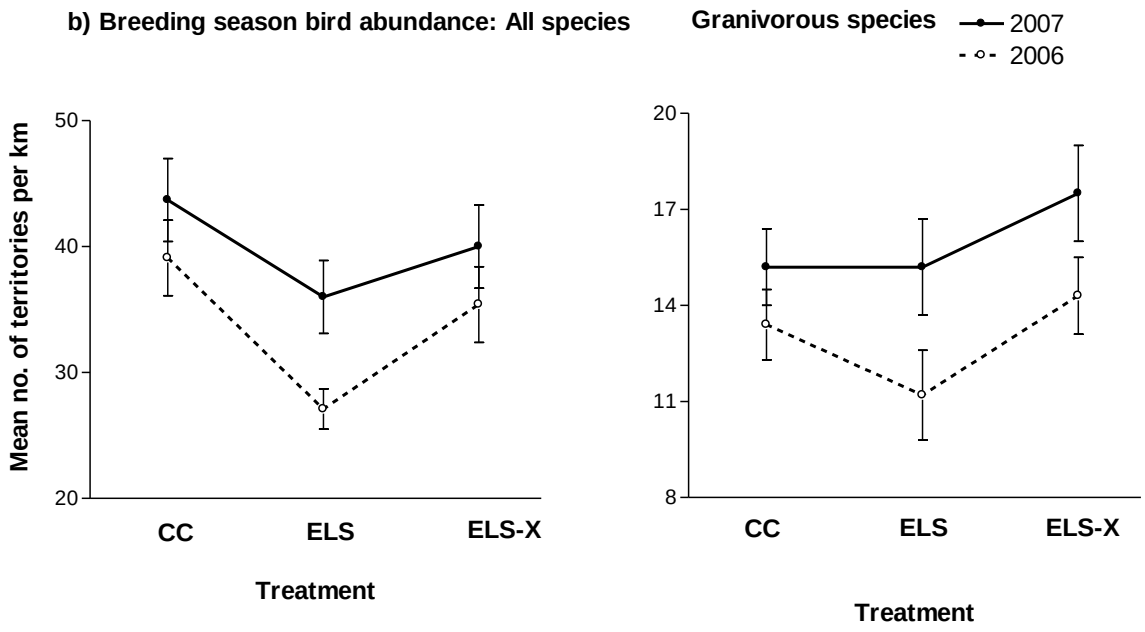


Fig. 5. Differences in bird abundance (for all species combined and granivorous species) between treatments in the winter (a: mean number of birds per km of transect) and breeding season (b: mean number of territories per km of transect) before (2005/06 and 2006) and after (2006/07 and 2007) bird food patch establishment. CC = Cross Compliance, ELS = Entry Level Scheme, ELS-X = Entry Level Scheme Extra; error bars show ± 1 se.


a) Winter bird abundance: All species
Granivorous species
2006/07
2005/06
Mean count per km
200
150
100
50
0
100
75
50
25
0
CC
ELS
ELS-X
CC
ELS
ELS-X


b) Breeding season bird abundance: All species
Granivorous species
2007
2006
Mean no. of territories per km
50
40
30
20
20
17
14
11
8
CC
ELS
ELS-X
CC
ELS
ELS-X
Treatment
Treatment

.

12.6 Published paper: " Performance and effectiveness of winter bird food patches established under Environmental Stewardship: Results from the Hillesden experiment"

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ABSTRACT

The Hillesden experiment is a farm-scale study evaluating the performance of options under Environmental Stewardship. We describe bird usage of winter seed patches (20 patches; three seed mixes) in relation to seed depletion and variation between individual patches. Seed retention declined exponentially in all three mixes; 50% depletion occurred by late November, reaching 80-90% before mid January. In mid winter, Fodder Beet retained more seed (*c.* 80%) than Millet, Kale, Fodder Radish and Triticale (20-40%). Bird numbers peaked in December/early January (seed depletion 70-90%), but declined rapidly in late January coinciding with seed exhaustion. Seed yields varied between patches (minimum < 1% of maximum). If all patches had performed at the maximum, yield would have increased by about 64%. Bird counts also varied greatly between patches, but trends with seed yield were positive. At a farm-scale, winter bird abundance was significantly greater (granivorous species + 415%) when patches were available.

INTRODUCTION

Over the last 40-50 years agricultural intensification has been responsible for large declines in many species of farmland birds and especially those most closely associated with cropland (Fuller *et al*., 1995; Chamberlain *et al*., 2000; Robinson & Sutherland, 2002). Amongst other factors, reductions in over-winter survival have been linked to reductions in winter food supplies for birds, for example due to total loss of over-winter stubbles to autumn cultivation (Wilson *et al*., 2009) and reduction in the quality of surviving stubbles (and other habitats) as foraging substrates due to increased use of pesticides (Potts, 1986; Wilson *et al*., 1999). Agri-environment schemes comprise a major component of UK Government policy designed to restore and enhance semi-natural habitat in farmland with the overall aim of reducing the loss of biodiversity and in particular to reverse the decline in farmland birds (Swash *et al*., 2000; Vickery *et al*., 2004). Environmental Stewardship (ES), made generally available to all farmers in England in 2005, is the most recent such scheme. Within ES, an option providing patches of dedicated winter bird food is designed specifically

to address the problem of over-winter survival. There is no doubt that birds use such food patches (Boatman & Stoate, 2002; Stoate *et al*., 2004), but there is as yet limited evidence that such use translates into increased survival and hence the desired response of population growth (Peach *et al*., 2001; Gillings *et al*., 2005). Indeed, the latest trend in the Farmland Bird Indicator (FBI), one of 18 UK Biodiversity Indicators averaging the population trends of 19 species of farmland birds, is downwards (JNCC, 2009). Furthermore, several studies (Siriwardena & Anderson, 2007; Siriwardena *et al*., 2008; BTO, 2009) have identified late-winter seed depletion (the so-called "hungry gap") in food patches as a potential cause of failure in their ability to deliver population recovery; birds may simply die later in the winter when the resource runs out.

The Hillesden Experiment, established in 2005/06 and scheduled to run for five years, is a farm-scale study designed to demonstrate and evaluate the cost-effectiveness of ES options to enhance farmland biodiversity. The work is wide-ranging and considers a number of taxa including soil micro and macro fauna, plants (arable weeds, hedgerow species), pollinators (bumblebees, solitary bees and their parasites), butterflies, moths, other invertebrates, small mammals and birds. Here we describe bird responses to the provision of dedicated winter seed patches in relation to seed depletion and variation in the performance of individual patches.

METHODS

The site is located on about 1000 ha of lowland arable farmland near Buckingham (51^{o}57' N, 1^{o}00' W) in central England. It is a typical heavy-land farm with a simple rotation of autumn-sown winter wheat, oil-seed rape and field beans. It is a randomised block experiment with five replicates of three treatments, each treatment being applied to contiguous areas of about 75 ha, with no gaps between the blocks. The control treatment (Cross Compliance) has no bird food patches while the other two have one patch (Entry-level) and three patches (Entry-level Extra) respectively making a total of 20 patches (i.e. Entry-level: 1 x 5 + Entry-level Extra: 3 x 5). The five Entry-level patches are 0.25 ha and the

Entry-level Extra patches are 0.5 ha making a total patch area of 8.75 ha. Patches were first sown in the spring of 2006 and thus were first available in the winter of 2006/07. Each of the Entry-level Extra patches comprised a different seed mixture hereafter referred to as Biennial, Deluxe and Bumblebird, while all the Entry-level patches were Biennial. The three seed mixes were Biennial: 70% Triticale (wheat/rye hybrid), 14% Kale *Brassica oleracea*, 14% Quinoa *Chenopodium quinoa* and 2% Fodder Beet *Beta vulgaris*; Deluxe: 40% Triticale, 20% White Millet *Echinochloa esculenta*, 20% Buckwheat *Fagopyrum esculentum*, 10% Quinoa and 10% Fodder Radish *Raphanus sativus*; Bumblebird: 40% Triticale, 15% White Millet, 15% Dwarf Sunflower *Helianthus annuus*, 10% Quinoa, 10% Fodder Radish, 7% Sweet Clover *Melliotus officinalis* and 3% Borage *Borago officinalis*. As the name implies, the Biennial mix produced seed over two years, the second winter seed crop being supplied by the Kale and Fodder Beet components. Thus in the winter of 2006/07 the Biennial patches were in their first year with the Kale and Fodder Beet seeding in the following winter of 2007/08. In practice, one of the Entry-level Biennial patches failed to establish and was resown for the winter of 2007/08 with the Deluxe annual mix.

The maximum seed crop for each patch was estimated from samples collected in September 2007. In each patch, five quadrats (1 x 0.5.m) were sampled at random across the whole patch, but avoiding the first 3 m from the edges. All seed, from both sown and unsown, i.e. weed, species, from all plants rooted within the quadrats was collected for each species. Seeds were separated from the chaff, counted, dried at 80^{o}C for 24 hours and then weighed. Seed crop data were not available for one Deluxe patch.

Seed depletion was measured on the 15 Entry-level Extra patches during the winter of 2007/08. Three individual plants of each of Kale, Fodder Beet, Fodder Radish and White Millet and three groups of Triticale stems were selected at random in well-grown parts of each patch and marked for relocation. Measurement of depletion varied between species due to their different morphologies, which affected the ease and precision of handling and counting, and to the ways in which they were attacked by the birds. Thus the following were recorded: Kale, the number of unopened seed pods; Fodder Beet, the number of seed pods; Fodder Radish, the number of totally intact, i.e. unpecked, seed pods; White Millet, the numbers of racemes; Triticale, the numbers of ears. Results were expressed as percentage seed retention. Fodder Radish was the only species with a starting value of 100%, all the

other species had initiated shedding at the date of the first visit. Assessments were made in October, November, January, February and March 2007/08 (Fig. 1).

Birds, identified to species, were counted on all 20 patches on each of six visits (October, November, December, early January, late January and February) in the winter of 2007/08. To minimise the risk of multiple counts of birds moving between patches, counts were made on the same day at approximately the same time by four observers each counting five patches. Patches were observed from a distance (if possible) and finally flushed by walking around the perimeter and through the centre. Winter bird abundance at a whole-farm scale was also estimated by censusing hedgerow transects on three visits (December, January and February) in the winters of 2005/06 (no patches) and 2006/07 (patches available). Transects (each *c.* 1-2 km long) were located along hedgerows within each treatment making a total of 15 transects and a total recording length of 16.15 km. Birds in the hedges, hedgerow trees and within 10 m of either side were recorded by spot mapping. The birds recorded in the winter were a mixture of residents, short and regional-scale migrants and long distance migrants in unknown proportions. Therefore, the overall winter abundance of each species was expressed as the number of individuals per kilometre per transect totalled across all three counts. Totals (per km) were then calculated for "all species" and for "granivorous species" (Chaffinch *Fringilla coelebs*, Linnet *Carduelis canabina*, Yellowhammer *Emberiza citronella*, Reed Bunting *Emberiza schoeniclus*, Tree Sparrow *Passer montanus*). Differences in bird abundance per km between the two winters were compared at the whole-farm scale using paired *t*-tests and the 15 transects as replicates.

RESULTS

Seed retention showed a pattern of exponential decline in all three patch types with 50% depletion occurring towards the end of November (Fig.1). Thereafter the biennial mix retained

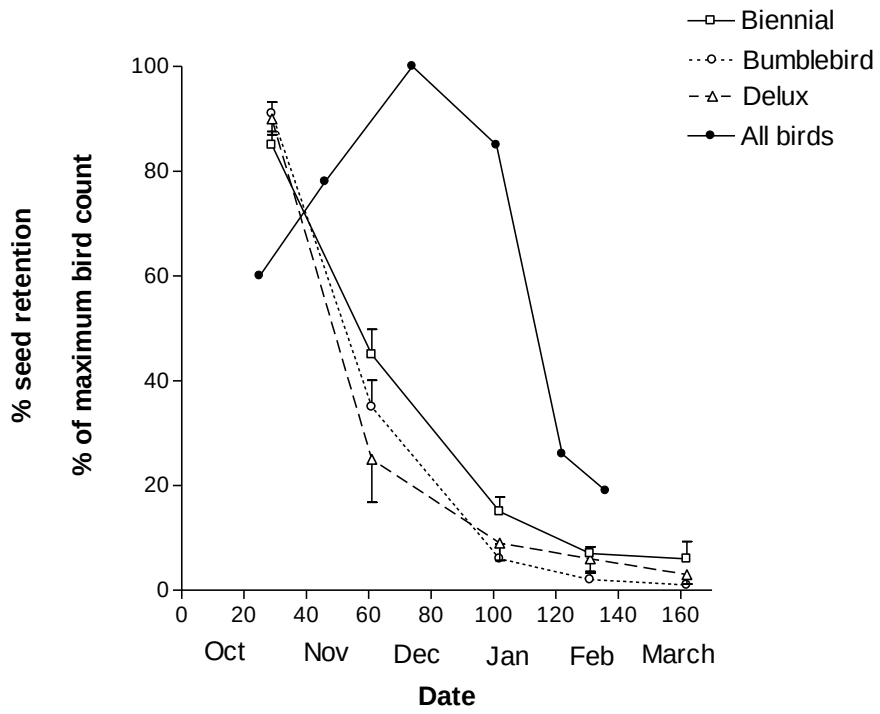

Fig. 1. Mean seed retention (%) by patch type in relation to bird use (% of maximum count for all species combined across all patches). October 1st = 1. For clarity, standard errors for seed retention are shown above the line for Biennial and Bumblebird and below for Deluxe.

more seed than either of the two annual mixes, but the overall difference was relatively small (maximum *c.* 20%, but usually c. 10-20%). By January 1st, mean seed depletion across all mixes was about 80% and exceeded 90% before mid January.

The greater retention of the biennial mix was due to Fodder Beet which retained some 80% of seed at the end of November compared to *c.* 20-40% for Millet, Kale, Fodder Radish and Triticale (Fig. 2). The pattern of depletion for Kale and Millet was very similar. Bird numbers on the patches reached a peak in December and early January (Fig. 1) when average seed depletion was c. 70-90% and Fodder Beet offered the most remaining seed (30-60% retention). Bird numbers then declined rapidly to *c.* 10% of maximum counts in late January and February.

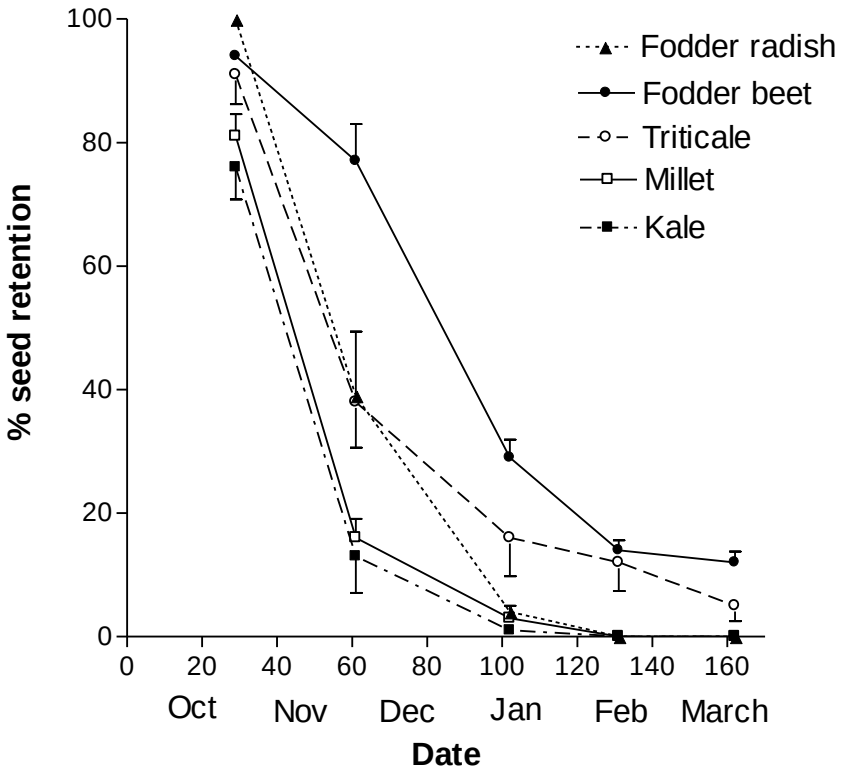

Fig. 2. Mean seed retention (%) by seed type. October 1st = 1. For clarity, standard errors are above the line for Fodder Beet, Fodder Radish, and Millet, and below for Triticale and Kale.

Seed production showed large variation between individual patches, both within patch types (Fig. 3) and across all patch types (Fig. 4). For example, the least productive Biennial mix patch produced about 90% less than the most productive one. It was estimated that if all patches had performed as well as the most productive within each type, then the total yield of sown species could have been increased by 64%. There was a negative correlation between the seed yields (kg per patch) of unsown and sown species (across all patches, corrected for patch area, r = -0.525, P = 0.021, n = 19); this relationship became stronger (r = -0.702, P = 0.001, n = 18) following the removal of one outlier (the seed yield of unsown species in this patch exceeded the mean for all the other patches by 7 SD). In general, sown seed yields exceeded unsown seed yields in the Biennial patches, whereas unsown yields tended to exceed those of sown species in the Deluxe and Bumblebird patches (Range of

seed yields, Biennial: sown 33-284 g m^{-2}, unsown 0.8-17 g m^{-2}; Deluxe: sown <0.1-18 g m^{-2}, unsown 8-84 g m^{-2}; Bumblebird: sown 0.6-31 g m^{-2}, unsown 15-37 g m^{-2}).

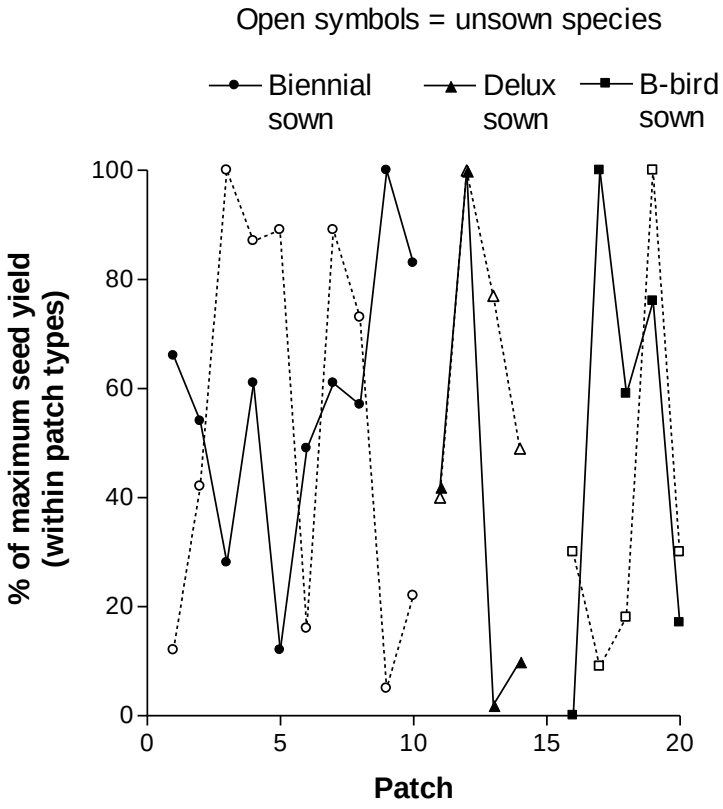

Fig. 3. Variation between individual patches in seed production (measured as g m^{-2} and corrected for patch area) expressed as percentage of maximum yield within each patch type.

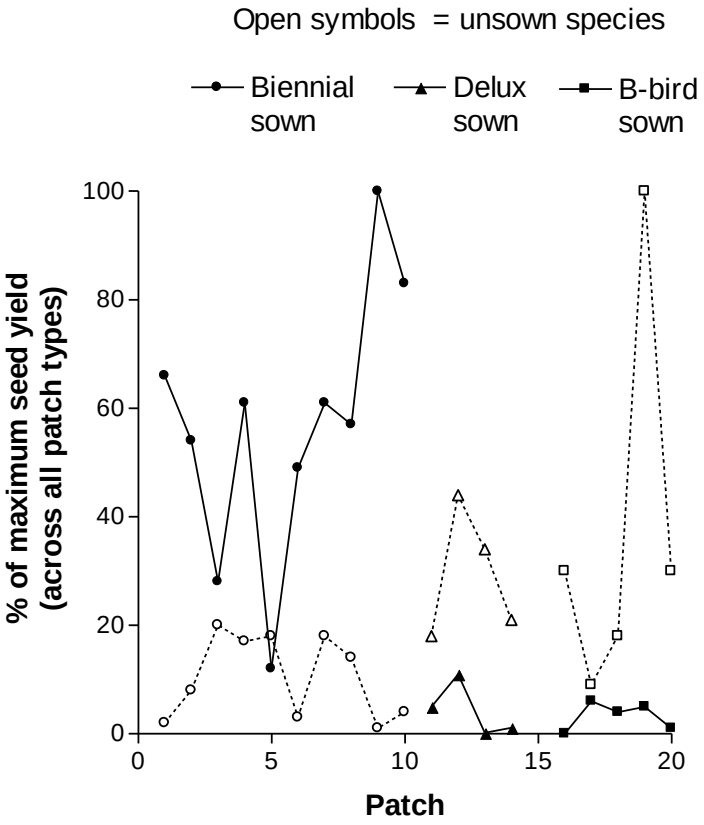

Fig. 4. Variation between individual patches in seed production (measured as g m^{-2} and corrected for patch area) expressed as percentage of maximum yield across all patch types.

Bird counts also varied greatly between patches (Fig. 5). Due to the large variation between patches in both seed production and bird counts, relationships between these two variables were seldom significant, but the trends were positive. A significant relationship was found between bird numbers and total seed yield (sown plus unsown species, kg per patch) per patch in November 2007 (Nov. bird count = 10.5 + 0.0776 total seed wgt, r^2 = 40%, P = 0.004). Using this relationship and the mean total seed yield per patch of 414.2 kg, if the seed yield of all patches had been as good as the above estimation of "best performance" (i.e. an increase in total seed production of 64%), this would predict an increase in bird numbers of 48%. The numbers of birds recorded at the whole-farm scale, using the 15 hedgerow transects as replicates, were greater in the winter of 2006/07, when the bird food patches were available, than in the previous winter of 2005/06 before the patches were established (Mean bird abundance per km of transect, All Species, without patches: 52.4, SE = 3.9, with patches: 118.5, SE = 9.6 [+126%], paired t-test P < 0.001;

Granivorous Species, without patches 9.9, SE = 1.4, with patches: 51.0, SE = 8.6 [+415%], paired t-test $P < 0.001$).

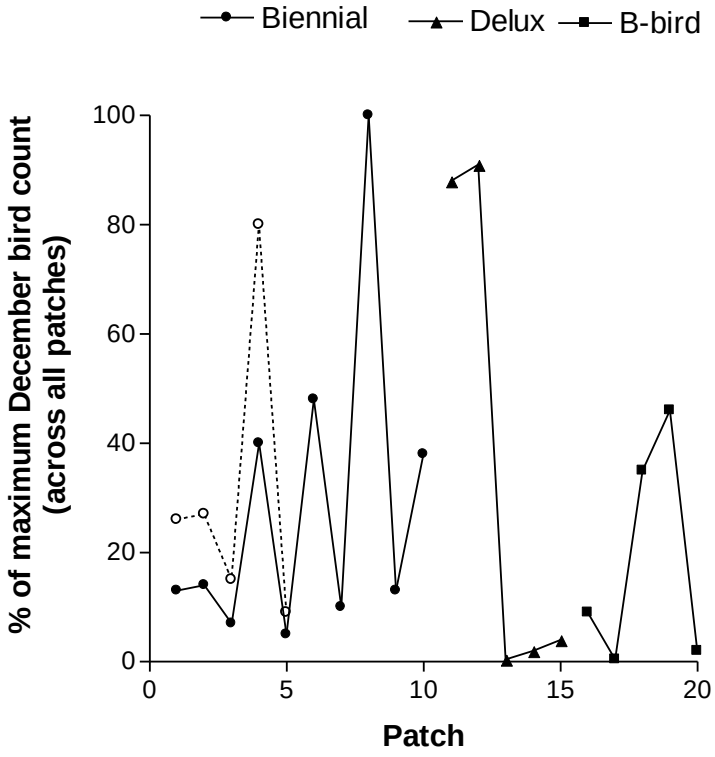

Fig. 5. Variation in bird usage of individual patches expressed as the percentage of the maximum bird count across all patches in December. The December count was selected because bird numbers on the patches were maximal in this month (see Fig. 1). Values corrected for patch size (patches 1-5 = 0.25 ha) are shown by open symbols and the dotted line.

DISCUSSION

Agricultural intensification has reduced winter food supplies for birds by a number of mechanisms including whole-sale habitat loss, conversion of winter stubbles to autumn cultivation and reductions in foraging habitat quality due to drainage and pesticide use (Wilson *et al*., 2009). Late winter (*c.* February to March) has been identified as the so-called

"hungry gap" when depleted food supplies have yet to be replenished and when various over-winter agri-environment habitats, including seed patches, may be ploughed (Siriwardena *et al*., 2008). The latter loss of foraging habitat will have been exacerbated by the withdrawal of set-aside in 2008. Seed loss in the bird food patches at Hillesden was consistent with this pattern of potential late winter food shortage, mean seed depletion across all patch types exceeding 90% by mid January. The coincident rapid drop in bird numbers on the patches in late winter was also indicative of dwindling food supplies. However, the significant increase in the numbers of birds, especially seed-eaters, present on the farm in general in the winter when the seed patches were available suggested that, despite problems with depletion, the patches did provide a valuable winter food resource. The challenge is to extend the "life-time" of this resource into February and March. The density of breeding territories of granivorous species was significantly greater in 2007 (patches present) than in 2006 (no patches) (16 territories per km of transect versus 13, Hinsley *et al*., unpubl. data) suggesting an additive positive influence on breeding population size, but more work is required to confirm a sustained trend.

Seed retention will depend on a number of factors including simple abundance of the resource, characteristics of the plant species and bird seed preferences. Increasing patch size and hence the size of the seed crop may feed birds for longer and/or simply increase the numbers of birds exploiting the resource. Although birds will move widely in winter in search of food (Siriwardena *et al*., 2006), the provision of sufficient resources to meet regional population demand also has potential to prolong the life-time of the seed supply. Such an approach would require uptake of bird food options within ES at a national scale. In addition to increasing patch size, the results at Hillesden showed that increasing the seed yield per patch had great potential for improvement; if all the patches had performed as well as the "best" patches, total seed production could have been more than doubled, with the potential to feed twice as many birds. There is a tendency for bird food patches (and other agri-environment habitats) to be located on the least productive land (e.g. areas prone to flooding, shading, rabbit damage etc.) and this, coupled with repeated use over several years, can result in poor establishment and yields. The negative correlation between the seed weights of sown and unsown, i.e. "weed", species in the patches was indicative of this effect. Although weed species may supply some resources, they do not substitute for sown species in terms of total abundance; there was a significant positive correlation (r =

0.982, $P < 0.001$, $n = 19$) between total seed yield and sown seed yield, but only a non-significant negative trend ($r = -0.357$, $P = 0.133$, $n = 19$) with unsown seed yield.

Plant characteristics influence seed retention, for example, Kale pods shed their seed explosively when touched, but Fodder Radish pods must be pecked opened to extract the seed. Seed size and ease of extraction from the plant will also interact with bird morphology and foraging behaviour (Wilson *et al*., 1999; Holland *et al.*, 2006); in general, smaller-billed species prefer smaller seeds. Thus at Hillesden, Linnets were commonly observed feeding on Kale whereas Fodder Beet was more commonly used by Greenfinches *Carduelis chloris* and Chaffinches. Triticale offers large seeds which should be attractive to buntings and the larger finches, but they may find extracting seed from the solitary ears difficult; some intact Triticale ears were present in the patches in February (M. Novakowski, pers. obs.). However, Rooks *Corvus frugilegus* were adept at removing whole ears which were then taken to the ground to remove the seed and probably accounted for the rapid depletion of Triticale in November. The physical structure of the patches also provided cover and access to damp ground and associated invertebrate food resources for species such Dunnock *Prunella modularis* and Song Thrush *Turdus philomelos*. Species feeding on the ground could also have access to shed seed, but would be in competition with other consumers such as small mammals.

Dedicated bird seed patches undoubtedly deliver food resources for birds in winter, but there is considerable scope to improve performance. In particular, achieving consistently high yields will require patches to be managed as carefully and effectively as cropland. Selection of seed mixes offering a range of seed sizes, nutritional qualities and retention characteristics, in relation to resource abundance, i.e. the number, type and size of patches, needs to be considered at a scale consistent with bird winter ranging behaviour. This could be of the order of 100s or 1000s of hectares and would require application/organisation at a national scale.

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REFERENCES

Boatman N D, Stoate C. 2002. Growing crops to provide food for seed-eating farmland birds in winter. *Aspects of Applied Biology* 67:229-236.

BTO. 2009. http://www.bto.org/news/news2009/janfeb/hungry_gap.htm, accessed May 2009.

Chamberlain D E, Fuller R J, Bunce R G H, Duckworth, J C, Shrubb, M. 2000. Changes in the abundance of farmland birds in relation to the timing of agricultural intensification in England and Wales. *Journal of Applied Ecol*ogy 37:771-788.

Fuller R J, Gregory R D, Gibbons D W, Marchant J H, Wilson J D, Baillie S R, Carter N. 1995. Population declines and range contractions among lowland farmland birds in Britain. *Conservation Biology* 9:1425-1441.

Gillings S, Newson S E, Noble D G, Vickery J A. 2005. Winter availability of cereal stubbles attracts declining farmland birds and positively influences breeding population trends. *Proceedings of the Royal Society of London Series B* 272:733-739.

Holland J M, Hutchison M A S, Smith B, Aebischer N J. 2006. A review of invertebrates and seed-bearing plants as food for farmland birds in Europe. *Annals of Applied Biology* 148:49–71.

JNCC. 2009. (http://www.jncc.gov.uk/page-4235, accessed May 2009.

Peach W J, Lovett L J, Wotton S R, Jeffs C. 2001. Countryside Stewardship delivers cirl buntings in Devon. *Biological Conservation* 101:361-373.

Potts G R. 1986. *The Partridge: Pesticides, Predation and Conservation*. London: Collins.

Potts, S.G., Biesmeijer, J.C., Kremen, C., Neumann, P., Schweiger, O., & Kunin, W.E. (2010) Global pollinator declines: trends, impacts and drivers. *Trends in Ecology & Evolution*, 25, 345-353.

Robinson R A Sutherland W J. 2002. Post-war changes in arable farming and biodiversity in Great Britain. *Journal of Applied Ecol*ogy 39:157-176.

Siriwardena, G.M., Calbrade, N.A. & Vickery, J.A. & Sutherland, W.J. 2006. The effect of the spatial distribution of winter seed food resources on their use by farmland birds. *Journal of Applied Ecology* 43:628-639.

Siriwardena G M, Anderson G Q A. 2007. How can agri-environment measures providing winter food for birds best deliver population increases? *Asects of Applied Biology* 81:117-126.

Siriwardena G M, Calbrade N A,Vickery J A. 2008. Farmland birds and late winter food: does seed supply fail to meet demand? *Ibis* 150:585-595.

Stoate C, Henderson I G, Parish D M B. 2004. Development of an agri-environment scheme option: seed bearing crops for farmland birds. *Ibis* 146 (Supple. 2):203-209.

Swash A R H, Grice P V, Smallshire D. 2000. The contribution of the UK Biodiversity Action Plan and agri-environment schemes to the conservation of farmland birds in England. In *Ecology and Conservation of Lowland Farmland Birds*, pp. 36-42. Eds N J Aebischer, A D Evans, P V Grice and J A Vickery. Tring: British Ornithologists' Union.

Vickery J A, Evans A D, Grice P V, Aebischer N J, Brand-Hardy R. Eds. 2004. Ecology and Conservation of Lowland Farmland Birds II: The Road to Recovery. *Ibis* 146 (Supple. 2).

Wilson J D, Evans A D, Grice P V. 2009. *Bird Conservation and Agriculture*. Cambridge: Cambridge University Press.

Wilson J D, Morris A J, Arroyo B E, Clark S C, Bradbury R B. 1999. A review of the abundance of invertebrate and plant foods of granivorous birds in northern Europe in relation to agricultural change. *Agrculture, Ecosystems and Environment* 75:13–30.